Section 22 Vermont

Phosphorous Transport and Attenuation

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New England Division

TECHNICAL REPORT

DEVELOPMENT OF A PHOSPHORUS-EUTROPHICATION MANAGEMENT STRATEGY FOR VERMONT: EVALUATING AVAILABLE PHOSPHORUS LOADS

by

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Research was conducted to (1) review current information on the availability of phosphorus derived from off-lake point and nonpoint sources for phytoplankton growth in receiving water bodies, (2) present eutrophication management strategies with respect to phosphorus control, and (3) provide guidance on approaches that should be used to develop updated phosphorus management strategy for the State of Vermont.

Pertinent information on phosphorus transport and availability was gathered from the literature, communication with other researchers in the field, and the authors' experience in developing phosphorus management strategies. (Cont'd on next page)

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It was found that, although significant amounts of phosphorus added to a river can be converted from forms that support algal growth ("available" phosphorus) to unavailable forms, the extent to which this occurs is highly river-specific. Field studies are recommended to determine the amount of algal-available phosphorus entering a river from point and nonpoint sources. Guidelines for conducting field studies are presented.

Eutrophication management strategies should include several activities such as: limiting phosphorus from all domestic wastewater treatment plants larger than 0.2 MGD to 1 mg/L total phosphorus; allowing exemptions to this rule if a community demonstrates that the restriction will not greatly improve water quality or will impose an excessive financial burden; and establishing controls on the storage and handling of animal manure.

FOREWORD

This research was conducted for the State of Vermont Department of Water Resources under the authority of Section 22 of Public Law 93-251, "Planning Assistance to States." The study was performed by the U.S. Army Construction Engineering Research Laboratory (USA-CERL) Environmental Division (EN) with cooperation from the New Jersey Institute of Technology under Project Order 85-C-033 (8 April 1985).

The U.S. Army Corps of Engineers, New England Division, acted as overall project manager, providing funding and technical review. Appreciation is extended to Martha Blake, USA-CERL, and R. Czaplinski and Eric Smeltzer, Vermont Department of Water Resources, for help in preparing the report.

Dr. R. K. Jain is Chief, USA-CERL-EN. COL Paul J. Theuer is Commander and Director of USA-CERL, and Dr. L. R. Shaffer is Technical Director.

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1 INTRODUCTION

Background

Like many states, Vermont has major problems with excessive fertilization (eutrophication) of its lakes and reservoirs. In particular, Lake Champlain is an important water body for which recreational use is being impaired by heavy algal growth in several of its bays.

The Vermont Department of Water Resources and Environmental Engineering has been conducting studies for several years to develop programs for managing eutrophication of this lake and other state waters. In 1977, Vermont adopted a eutrophication control program for Lake Champlain that focused on reducing the phosphorus input to the lake from domestic wastewater treatment plants discharging directly into the lake or to a nearby tributary. These treatment plants were required to upgrade their methods by including phosphorus removal to a 1 mg/L effluent P concentration.

"Offlake" domestic wastewater treatment plants and diffuse sources of phosphorus were exempted from these control programs, but the Department of Water Resources and Environmental Engineering now wants to learn if these exemptions are appropriate. Further, the department may want to implement phosphorus control programs for nonpoint sources. Of particular concern is how much phosphorus discharged to Lake Champlain tributaries reaches the lake in a form available to stimulate algal growth. A literature review on phosphorus interactions in the aquatic environment was completed in 1978;[1] however, it did not address the new issues. Therefore, the U.S. Army Corps of Engineers, New England Division, was asked to provide assistance through the Section 22 Program, "Planning Assistance to States." The U.S. Army Construction Engineering Research Laboratory (USA-CERL) was contracted to perform the study because of its experience in phosphorus management and modeling studies.

Objectives

The objectives of this study were to (1) review current information on the availability of phosphorus derived from offlake point and nonpoint sources for phytoplankton growth in receiving water bodies; (2) present eutrophication management strategies adopted in other watersheds with respect to phosphorus derived from offlake point and nonpoint sources; and (3) provide guidance on approaches that should be used to develop an updated phosphorus management strategy for the State of Vermont.

[1] D. W. Meals, Jr., and E. A. Cassell, *Phosphorus Interactions in the Aquatic Environment: A Survey of the Literature*, Report to the Water Quality Work Group of the Lake Champlain Basin Study (February 1978).

Approach

USA-CERL met with the Vermont Department of Water Resources and Environmental Engineering, the New England Division Army Corps of Engineers, and the New Jersey Institute of Technology to define specific concerns with reference to the availability of phosphorus (from offlake point and nonpoint sources) to support phytoplankton growth in Lake Champlain. Pertinent information on phosphorus transport and availability was gathered from the literature, communication with other researchers in the field, and the authors' experience in developing phosphorus management strategies for various water bodies worldwide.

Scope

This report addresses concerns specific to the state of Vermont Department of Water Resources and Environmental Engineering. However, principles and guidelines may apply to watersheds outside Vermont.

Mode of Technology Transfer

Information in this report provides the basis for an Engineer Technical Note (ETN) on phosphorus management strategies for dealing with offlake point and nonpoint sources.

2 CURRENT INFORMATION ON ASSESSING ALGAL-AVAILABLE PHOSPHORUS

The Meals and Cassell report is a comprehensive literature review of phosphorus behavior in aquatic environments, especially in terms of factors that influence phosphorus' role in eutrophication of natural water systems. It also covers the assessment of algal-available phosphorus.

Another review covers assessment methods for algal-available phosphorus in aquatic systems.[2] This review was prepared for an International Joint Commission conference devoted to developing phosphorus management strategies for lakes—in particular, the U.S.-Canadian Great Lakes. The long history of eutrophication in the Great Lakes and the former availability of funds for research on managing this problem led to many studies designed specifically to assess algal-available phosphorus in the Lakes' tributaries. The Lee et al. review covers this literature through the late 1970s. Because of severe curtailments in funding for Great Lakes research since then, little additional work on this topic has been done in the Great Lakes basin or elsewhere. However, the consensus at the July 1985 International Conference on Management Strategies for Phosphorus in the Environment, Lisbon, Portugal, was that the Lee et al. report still represents a comprehensive review of current information. Other work in progress or recently completed on algal-available P seems to support the Lee et al conclusions.

Phosphorus Availability in Aquatic Systems

To develop a strategy for managing phosphorus, it is essential to know which forms of phosphorus are important in aquatic systems, their availability for algal growth, and their conversion to unavailable forms. The rate at which phosphorus is cycled and exchanged between particulate, soluble inorganic and soluble organic forms also must be considered, as this often determines the rate of algal growth in a water body.[3]

Soluble Orthophosphate

A variety of phosphorus compounds is present in aquatic systems, only some of which are available to support algal growth. It has been found that soluble orthophosphate, properly measured using an ascorbic acid analytical procedure,[4] is readily available for algal growth. There are some problems with using this analytical procedure at very low levels of phosphorus (on the order of 1 μg/L P) due to interference from chemicals such as arsenic.[5] These chemicals can make the soluble ortho P concentration in the samples appear elevated. However, these potential errors in the analytical procedure are rarely significant in developing a phosphorus management strategy for a water body because such a strategy must focus on sources discharging phosphorus at concentrations considerably above a few μg/L.[6]

[2]G. F. Lee, R. A. Jones, and W. Rast, "Availability of Phosphorus to Phytoplankton and Its Implication for Phosphorus Management Strategies," *Phosphorus Management Strategies for Lakes* (Ann Arbor Science, 1980).
[3]R. G. Wetzel, *Limnology* (S. B. Saunders Co., 1975).
[4]American Public Health Association, *Standard Methods for the Examination of Water and Wastewater*, 16th ed. (1985).
[5]G. F. Lee, et al. (1980).
[6]G. F. Lee, et al. (1980).

In conducting soluble ortho P measurements, it is important to separate the soluble forms of phosphorus from particulate forms within about 24 hr of collection using membrane filters rinsed properly with acid. There will be some conversion of soluble ortho P to particulate forms, and vice versa, during this period. This conversion is not important since, with few exceptions, the same conversions most likely occur in the aquatic system being investigated. Further information on handling these samples and conducting the analyses is presented elsewhere.[7]

Condensed Phosphates

Forms of soluble phosphorus other than orthophosphate are present in aquatic systems; some of these are available for algal growth either directly or indirectly after hydrolysis and/or mineralization. Condensed phosphates such as pyro- and tripolyphosphate in detergent formulations are of particular concern. Studies have shown these compounds to have a fairly rapid hydrolysis which is promoted mainly by bacteria and appears to result in the formation of orthophosphate.[8] However, under controlled laboratory conditions, less algal growth (about 50 percent) was obtained in cultures using condensed phosphates as the P source than was obtained when the same amount of soluble ortho P was added to the culture.[9] Similar results have been reported from France.[10] The latter two studies have shown that, during hydrolysis of the condensed phosphate, some phosphorus is converted to forms not immediately available for supporting algal growth. It appears that part of the condensed phosphate is converted to phosphorus-oxygen-carbon (POC) or phosphorus-carbon (PC)-bonded organic phosphorus compounds that are not readily (i.e., within 1 to 2 weeks) converted by bacteria in the study systems to algal-available P. Mild acid hydrolysis of the sample did convert these chemicals to orthophosphate.[11] No information is available on whether, with extended periods of incubation (i.e., longer than several weeks), bacteria would be able to convert the unavailable P forms to available forms.

These results have important implications in assessing how effective detergent phosphate bans are in reducing algal growth in natural waters. Detergent phosphate bans such as those adopted in Vermont are likely to have limited impact on most water bodies' eutrophication-related quality because the conditions necessary for a major improvement usually are not present. If, as the experimental data suggest, half the condensed phosphate from detergents is converted to permanently unavailable forms during hydrolysis in domestic wastewater treatment plants and in surface water systems, it would again be rare that phosphate bans would affect eutrophication-related water quality; these bans would remove such a small part of the algal-available P from domestic wastewaters that they would be ineffective in controlling eutrophication. It should be noted that, thus far, no documented case has been observed in which a detergent phosphate ban improved water quality. This statement is in accord with results from others in the field.[12]

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[7] G. F. Lee, et al. (1980).
[8] G. F. Lee, et al. (1980).
[9] N. L. Clesceri and G. F. Lee, "Hydrolysis of Condensed Phosphates I: Non-Sterile Environment," *Air and Water Pollution*, Vol 9 (1965).
[10] D. Villesot, M. Jaubert, C. Laval, and P. Haignere, "Bio-Availability of Phosphorus in Natural Waters," *Proceedings, International Conference on Management Strategies for Phosphorus in the Environment* (Selper Ltd., London, 1985).
[11] N. L. Clesceri and G. F. Lee.
[12] A. W. Maki, D. B. Porcella, and R. H. Wendt, "Impact of Detergent P Bans on Receiving Water Quality," *Water Research*, Vol 18 (1984).

Phosphorus Derived From Wetlands

Drainage from wetlands is another source of dissolved phosphorus in aquatic systems.[13] One study showed that a large amount of the phosphorus in water discharged from Wisconsin wetlands was in a soluble organic form.[14] No work was done on the availability of this phosphorus to support algal growth. However, it is likely that at least part of it will be mineralized to orthophosphate and thereby support algal and other aquatic plant growth in aquatic system. This area needs further investigation.

Other research determined inorganic phosphorus uptake rates by aquatic macrophytes in summer and winter.[15] Plant uptake accounted for 12 to 73 percent of the phosphorus removed from the water column. A separate study found that wetlands adjacent to White Clay Lake with a predominantly agricultural watershed retained phosphorus at a rate of 2.6 g/m^2 of watershed area/yr over a 9-yr study period.[16] This result is significant when compared with monitored sediment and P loading.

The State of Vermont Department of Water Resources and Environmental Engineering wants to assess the impact of phosphorus in wetland drainage on eutrophication of downstream reservoirs and lakes. In addition to the standard algal assays, it is important that such assessments for any particular wetland system use algal bioassay techniques (Chapter 3) to test the phosphorus' algal-availability in the drainage.

Particulate Phosphate

A primary mechanism by which algal-available phosphorus is converted to unavailable forms is in the formation of particulate P compounds. These compounds can be formed in several ways, one of the most important of which is through incorporation of soluble orthophosphate into plant tissue. Upon the plant's death, part of the phosphorus that was taken up as soluble ortho P remains with the organic detritus as particulate P. Much of this detritus eventually is mineralized by bacterial action, with the associated phosphorus becoming available again to support aquatic plants (algae and macrophytes). Although most phosphorus taken up by aquatic plants is mineralized to form orthophosphate, a small but readily measurable fraction is converted to refractory P forms that will never again become available to support algal or macrophyte growth.[17] This finding was confirmed by another study, which found that about 90 percent of the phosphorus taken up by phytoplankton can be mineralized to soluble orthophosphate.[18] Other work showed that about 1 to 5 percent of the phosphate taken up by algae is not returned to the biochemical cycle.[19] This means substantial available

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[13]G. F. Lee, et al. (1980).

[14]G. F. Lee, E. Bentley, and R. Amundson, "Effect of Marshes on Water Quality," *Ecological Studies 10, Coupling of Land and Water Systems* (Springer-Verlag, 1975).

[15]K. R. Reddy, "Nutrient Transformations in Aquatic Macrophyte Filters Used for Water Purification," presented at the Water Reuse Symposium III, San Diego (August 1984).

[16]L. A. Persson, et al., "Evaluation of Sediment and Phosphorus Management Practices in White Clay Lake Watershed," *Water Resources Bulletin*, Vol 19, No. 5 (1977).

[17]G. F. Lee, et al. (1980).

[18]J. V. DePinto, T. C. Young, J. S. Bonner, and P. W. Rodgers, "Microbial Recycling of Phytoplankton Phosphorus," *Canadian Journal of Fisheries and Aquatic Science* (1985).

[19]H. L. Golterman, "Natural Phosphate Sources in Relation to Phosphate Budgets: A Contribution to the Understanding of Eutrophication," *Water Research*, Vol 7 (1973).

P could be lost over a 200-day growing season; with growth cycles of 5 to 10 days, the amount taken up by algae could be 20 to 100 percent of the available phosphorus. Therefore, with multiple passes through aquatic plant systems, such as would occur in the transport of phosphorus in long, slowly moving streams and in the nearshore waters of lakes and reservoirs, most available P forms added to the system will be converted to unavailable forms.

Calcium Phosphate

In addition to the biotic processes just discussed, several abiotic processes can convert algal-available P to unavailable forms. One of the most important abiotic mechanisms is the direct precipitation of phosphorus as a calcium phosphate species. One paper dealing with this topic investigated the forms of phosphorus that develop in calcareous soils.[20] The insoluble forms of phosphorus behaved as octacalcium phosphate and dicalcium phosphate anhydrous species. Other reports have found that hydroxy apatite species form on the surface of calcium carbonate crystals.[21] In contrast, little evidence for hydroxy apatite formation was noted in a different system studied.[22] It appears that formation of calcium phosphate species such as hydroxy apatite is controlled kinetically by nucleation rates, since supersaturated waters have been reported by several investigators.[23]

Although the exact forms of phosphorus incorporated into calcium phosphate species will vary depending on the system, it is clear that such species will form in systems that experience calcium carbonate precipitation. The formation of calcium carbonate species in aquatic systems would tend to convert algal-available forms of phosphorus, such as soluble orthophosphate, to unavailable forms. Thus, aquatic systems that are precipitating calcium carbonate might be expected to be less productive than noncalcareous systems. However, one found no difference in the overall productivity of lakes and reservoirs that had elevated concentrations of calcium and alkalinity compared with low-hardness, low-alkalinity waters.[24] It therefore appears that, though calcium phosphate species form in aquatic systems and help make available P forms unavailable, the overall significance of this process in controlling a water body's productivity is small. A possible exception would be water bodies that precipitate massive amounts of calcium carbonate such as marl lakes, in which the water becomes a milky white due to precipitation. The productivity of some marl lakes studied is less that what would be expected based on the nutrient loads.[25] It is important to stress, however, that this situation appears to be unusual.

If a system contains hard water that precipitates calcium carbonate and if soluble orthophosphate were added to water at a point allowing it several days transit time before reaching a larger water body, would it be delivered in an algal-available form?

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[20]G. A. O'Connor and K. L. Knudtsen, " Phosphorus Solubility in Sludge-Amended Calcareous Soils," *Proceedings, International Conference on Management Strategies for Phosphorus in the Environment* (Selper, Ltd., London, 1985).

[21]W. Stumm and J. J. Morgan, *Aquatic Chemistry: An Introduction Emphasizing Chemical Equilibria in Natural Waters*, 2nd ed. (John Wiley and Sons, 1981).

[22]G. A. O'Connor and K. L. Knudtsen.

[23]G. F. Lee, et al. (1980).

[24]W. Rast and G. F. Lee, *Summary Analysis of the North American (U.S. Portion) OECD Eutrophication Project: Nutrient Loading-Lake Response Relationships and Trophic State Indices*, EPA-600/3-78-008 (1978)

[25]R. G. Wetzel.

No data are available to answer this question. This might be expected to be the case; however, site-specific evaluations must be made to determine if this factor greatly influences the amount of algal-available P reaching a lake or reservoir after being discharged to a river at some distant point.

One of the first steps to take in this evaluation is to determine if calcium carbonate tends to precipitate from the river. Also, the tendency for calcium phosphate species to precipitate should be examined. Established relationships[26] can be used to determine the potential for calcium carbonate precipitation, whereas the relationships published elsewhere[27] help show the tendency for calcium phosphate species to form.

Iron and Aluminum Phosphate

It is likely that both iron and aluminum phosphate species form in aquatic systems. In fact, adding iron or aluminum salts to domestic wastewaters is a common method of phosphorus removal. The iron and aluminum flocculants remove soluble and particulate phosphorus from the water column and incorporate the phosphorus into the sludge. The primary mechanism for removal appears to be sorption. Treating a whole lake with iron or aluminum salts has been found to be effective in improving the lake's eutrophication-related water quality.[28] In alum-treated systems, no significant phosphorus was released from the sediment in the years after treatment. Iron-treated systems, however, would be expected to release phosphorus whenever the waters become anoxic, enabling the ferric iron to be reduced to the ferrous form. Because the aluminum phosphate species will permanently bind phosphorus into the water's sediments, alum treatment often is favored over iron salts addition.

Alum also is used to tie up phosphate in combined sewer overflows and urban stormwater drains; it is added to the waters at the point where the overflow or drainage enters the aquatic system. This approach has considerable merit for controlling phosphorus from sources that are difficult if not impossible to control otherwise. Along with other remedial measures, alum has been applied successfully to control internal phosphorus loading in a small seepage lake.[29]

Although adding alum or iron to a lake or river can convert large amounts of algal-available phosphorus to unavailable forms, the significance of this process in aquatic systems without added external sources of iron or aluminum is unknown. The sorption capacity for phosphates of aged iron and aluminum hydrous oxide precipitates is considerably less than that of freshly precipitated chemicals.[30] About the only way such freshly precipitated chemicals can be introduced into natural waters without external addition is through the scour of bottom sediments typically occurring with elevated river flow or storms.[31] Under these conditions, ferrous iron in the anoxic sediments brought into the oxic water column is oxidized to ferric iron, which in turn precipitates as ferric hydroxide, incorporating some phosphate into the ferric hydroxide floc. Studies have

[26] American Public Health Association (APHA).
[27] G. A. O'Connor and K. L. Knudtsen.
[28] G. F. Lee, "Eutrophication," *Transactions of the Northeast Fish and Wildlife Conference* (1973); G. F. Lee, et al. (1980).
[29] P. J. Garrison, et al., *Lake Restoration, a Five Year Evaluation of the Mirror and Shadow Lakes Project, Waupaca, WI,* EPA-600/3-83-010 (March 1983).
[30] G. F. Lee, et al. (1980).
[31] G. F. Lee and R. A. Jones, "Application of the OECD Eutrophication Modeling Approach to Estuaries," *Estuaries and Nutrients* (Humana Press, 1981).

shown that a suspension of U.S. waterway sediments usually releases appreciable phosphorus, which is incorporated into particulate forms upon contact with dissolved oxygen.[32] The ferric hydroxide formed from ferrous iron's oxidation in the sediments sorbs the phosphate released from the sediments.

Particulate Forms

In addition to sorption on iron and aluminum hydrous oxides, some phosphate will sorb on other forms of particulate matter present in natural waters, such as organic detritus, clay minerals, and detrital minerals. Little is known about these reactions except that they occur; their significance in converting algal-available phosphorus to unavailable forms is largely unknown. Also of note is that in systems for which organic matter is mineralized, the sorbed phosphate may be converted to available forms. The soluble orthophosphate release when organic phosphate compounds mineralize may never appear in the water column; instead, it may reprecipitate immediately in some other form of particulate phosphorus.

Impact of River Transport on Phosphorus Availability

Riverine systems' tendency to convert algal-available phosphorus to unavailable forms is well known. However, the factors governing which forms of phosphorus are in aquatic systems are not understood well enough at present to predict reliably how much phosphorus added to a riverine system will be converted to unavailable forms.

One study has measured total and soluble phosphorus concentrations at 26 sampling stations along the Sandusky River in northwestern Ohio.[33] Samples were taken under nonstorm flow conditions during summer 1974 at Sandusky River stations downstream from three different sewage treatment plants (all without phosphorus removal). A rapid decline in total and soluble phosphorus concentrations was seen downstream from these plants (Figure 1). Cumulative streamflow and phosphorus flux were tabulated and showed that, for one plant, the phosphorus flux was less than the phosphorus loading from the treatment plant 55 percent of the time. This finding led to the conclusion that some phosphorus transformation and deposition was occurring downstream from the sewage treatment plant. Bio-availability (measured by extraction) of particulate phosphorus in watersheds with point sources was no different than in those without point sources. During storms, soluble reactive phosphorus exported to the lake was similar in watersheds with and without point sources. This result further supports the notion that

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[32]G. F. Lee, R. A. Jones, F. Y. Saleh, G. M. Mariani, K. H. Homes, J. S. Butler, and P. Bandyopadhyay, *Evaluation of the Elutriate Test as a Method of Predicting Contaminant Release During Open Water Disposal of Dredged Sediment and Environmental Impact of Open Water Dredged Material Disposal, Vol II: Data Report*, Technical Report D78-45 (U.S. Army Waterways Experiment Station [WES], 1978a); R. A. Jones and G. F. Lee, *Evaluation of the Elutriate Test as a Method of Predicting Contaminant Release During Open Water Disposal of Dredged Sediment and Environmental Impact of Open Water Dredged Material Disposal, Vol I: Discussion*, Technical Report D78-45 (WES, 1978); R. A. Jones and G. F. Lee, "The Significance of Dredging and Dredged Material Disposal as a Source of Nitrogen and Phosphorus for Estuarine Waters," *Estuaries and Nutrients* (Humana Press, 1981).

[33]D. B. Baker, *Fluvial Transport and Processing of Sediment and Nutrients in Large Agricultural River Basins* (U.S. Army Corps of Engineers, Buffalo District, February 1982).

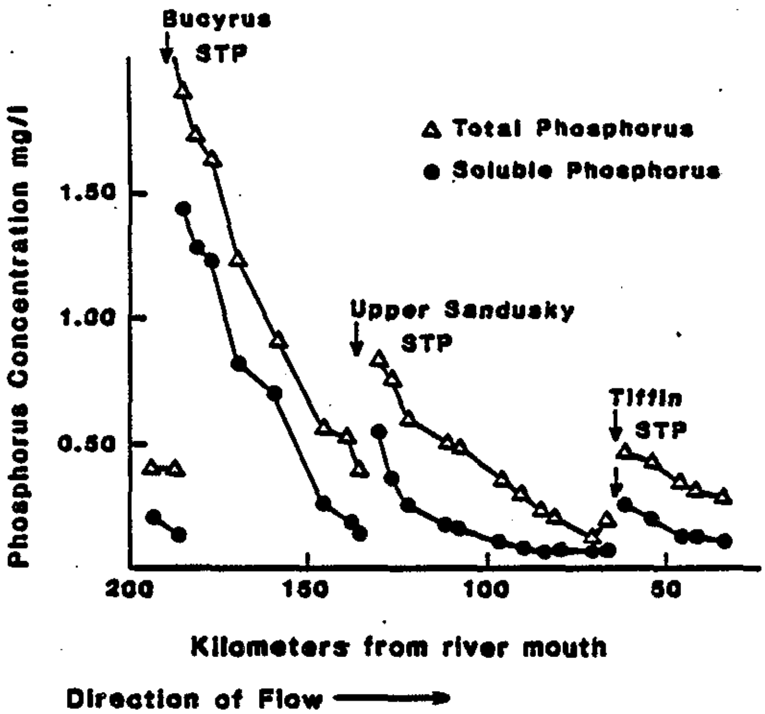

Figure 1. **Profile of mean phosphorus concentrations—Sandusky River** (Source: D. B. Baker, *Fluvial Transport and Processing of Sediment and Nutrients in Large, Agricultural River Basins* [U.S. Army and U.S. EPA, Corps of Engineers, Buffalo District, February 1982].)

point-source-derived phosphorus is transformed as it is carried downstream. However, some of this phosphorus may be resuspended and exported in particulate form during storms.[34]

Because previous investigations have not been conducted over the long term, it is still unknown whether the loss of soluble or total P in a river system represents a transitory conversion of phosphorus to particulate forms. It could be that these particulate forms, deposited in the riverbed under low flow conditions, are then scoured from the riverbed and release the P again. Some experience has indicated that even though soluble ortho P decreases rapidly below a point source discharge, such as a domestic wastewater treatment plant, this P would likely become available to support algal growth in downstream waters if the original decrease were the result of uptake by stream algae.

[34]D. B. Baker, *Fluvial Transport and Processing of Sediments and Nutrients in Large Agricultural River Basins*, Project Summary, USAEPA 600/53-83-054 (January 1984).

In hard-water systems from which phosphorus may be precipitated as a calcium phosphate species, it is highly likely that much, if not most, calcium phosphate will remain unavailable to support algal growth either in the river or in a water body receiving the river's discharges. The same situation probably exists for orthophosphate sorbed on clays or other mineral fractions. Further, some part of the phosphorus used by aquatic plants in the river system will be converted to forms of phosphorus that cannot be converted back to algal-available forms through bacterial mineralization of the plant material in slow-moving, sluggish streams.[35] However, as already noted, most phosphorus in aquatic plants will likely be converted to algal-available forms as part of the plant material's bacterial mineralization. In systems that have a rapid turnover of algae, multiple cycles through the algae could result in most added phosphorus being converted to unavailable forms during the growing season. The extent to which this occurs depends on the type of algae present and the system's characteristics. Another factor to consider is that large amounts of the aquatic plant-bound phosphorus may mineralize before the high spring flows that typically occur in temperate climates. If this happens, a greater percentage of the phosphorus originally discharged (used by plants and then mineralized to available forms) will be converted to unavailable forms as a result of the soluble phosphorus being sorbed onto the inorganic materials (e.g., clays, mineral fragments) transported during high spring flows.

Another mechanism by which phosphorus can be removed is incorporation into the sediments of reservoirs located on the river. For example, a lakewide average of 15 to 20 percent of the total phosphorus load from tributaries to Lake Michigan may be lost in lakes, bays, and impoundments.[36] In general, reservoirs that provide at least 2-weeks' hydraulic residence time during the summer will remove from 40 to about 75 percent of the phosphorus load added to a water body.[37] Part of this load is removed as the result of particulate phosphorus forms settling after entering the reservoir; most is removed by algal uptake and the transport of dead algae to the sediments. While some of this dead-algae-associated phosphorus will remineralize to soluble orthophosphate, much of the soluble ortho P formed in the reservoir's deeper waters will be sorbed or precipitated as inorganic phosphates. Therefore, it will be removed permanently from the water column and will not likely become available to support algal growth.

For river systems that tend to flood out of their normal channels each year and deposit part of their sediment loads in the flood plain area, a large part of the phosphorus may be lost from the river system and will not reach a downstream water body. The deposition of organic detrital phosphorus that could mineralize to orthophosphate in the river or downstream lake system in the flood plain can be a benefit in managing eutrophication of downstream water bodies. However, terrestrial plants can mobilize phosphorus from soils into forms at first unavailable to algae; when these plant remains are introduced into the river system and mineralized, the organic phosphorus deposited in the flood plain can be converted to algal-available forms again. A similar situation occurs in the shallow waters of lakes and reservoirs in which macrophytes pump algal-unavailable forms of phosphorus from the sediments through the plants to the water column. Thus, the death of macrophytes or flooded terrestrial plants that develop on the

[35] G. F. Lee, et al. (1980).

[36] T. M. Heidtke, et al., *Future U.S. Phosphorus Loadings to the Great Lakes: An Integration of Water Quality Management Planning Information* (U.S. Water Resources Council, 1979).

[37] G. F. Lee and B. A. Winkler, *Management of Water Quality in Lake Olathe*, Report to the City of Olathe, KS, Department of Civil and Environmental Engineering (New Jersey Institute of Technology, 1984).

reservoir sediments during periods of major reservoir drawdown can be important sources of phosphorus to the lake or reservoir.

This discussion should clarify why it is difficult to estimate, based on river characteristics, the amount of algal-available phosphorus discharge from a particular upstream source that will reach a water body as available P. Calcareous, slow-moving streams that have high inorganic turbidity tend to convert available forms of phosphorus to unavailable forms. In contrast, it is likely that rapid-moving, noncalcareous, relatively clear rivers will transport most of the added available phosphorus to a water body with little conversion to unavailable forms. Both types of rivers can be found in Vermont. This second case is true even under conditions in which large amounts of the added phosphorus are removed during the summer growing season through incorporation into aquatic plants within the river system.

Field studies are needed for a variety of situations to determine how much algal-available phosphorus added to a river system will be converted to unavailable forms. For example, mass balances of algal-available phosphorus should be measured at various locations on rivers over several annual cycles. Similarly, studies should be done to determine how much relatively unavailable P entering the river, such as forms associated with organic detritus, is converted to available forms in the river or in the water body receiving the river's discharge. These studies would eventually support some general estimates of an offlake phosphorus source's potential significance to water quality in the lake or reservoir receiving the phosphorus. Of particular concern in these studies is the reliable measurement of total P and soluble ortho P loads as well as the algal-available phosphorus measured by algal bioassay techniques.

3 FIELD STUDY APPROACH FOR ASSESSING AVAILABLE PHOSPHORUS

Overview

As Chapter 2 has suggested, it is not possible to reliably estimate the amounts of algal-available P based on simple chemical measurements. As a first approximation, algal-available P in agricultural, forest, and urban drainage can be estimated as the sum of the soluble ortho P and some fraction of the particulate phosphorus, all measured properly. Several investigations have found that about 20 percent of the particulate P in the sources studied is likely to become available for algal growth in lakes.[38] To use this value, the measurements must be made near the point at which the tributary source enters the lake. This 20 percent value also has been found for particulate matter in the discharge of alum-treated domestic wastewaters.[39] However, caution should be used in applying this value to waters derived from sources significantly different from those on which the value was determined.[40]

Work since publication of the review paper by Lee et al. has confirmed this value. For example, a long-term project is underway for assessing nonpoint-source phosphorus loads to parts of western Lake Erie (results not yet published). About 30 percent of the particulate P in the systems under study is hydroxide-extractable, which indicates this phosphorus would likely become available to support algal growth in downstream waters. Again, caution should be taken in using hydroxide-extractable P as a measure of algal-available P under conditions for which large amounts of phosphorus in the samples are organic. As already discussed, this form of phosphorus could become available through mineralization of organic matter.

The only known data on algal-available P in rivers that differ substantially with respect to the amount of available P in aquatic system particulate matter are for a region of South Africa (P. Groebbler, results not yet published). In this work, essentially all particulate P apparently was algal-available. The reason for this deviation is unknown. Care should be taken in applying the 20 percent particulate P figure to waters with heavy loads of particulate organic matter such as may occur in no-till farming areas or wetlands; mineralization of this particulate P could allow most of it to become available to support algal growth in receiving waters.

Algal Bioassays

The most reliable approach for assessing the amount of algal-available P in a water sample is to conduct algal bioassays.[41] This involves taking water samples at periodic intervals, such as weekly, over at least 1 year but preferably over several years. Total P should be measured (using persulfate digestion and ascorbic acid color development[42]) as well as soluble ortho P (on water filtered through a prerinsed 0.45 μ pore-size membrane filter within about 24 hr of sample collection, analyzed using ascorbic acid color

[38]G. F. Lee, et al. (1980).
[39]T. C. Young and J. V. DePinto, "Algal-Availability of Particulate Phosphorus From Diffuse and Point Sources in the Lower Great Lakes Basin," *Hydrobiologia*, Vol 54 (1982a).
[40]G. F. Lee, et al. (1980).
[41]G. F. Lee, et al. (1980).
[42]APHA.

development[43]). *Careful attention should be given to proper filtration and filter* prerinsing. Algal growth stimulation bioassays[44] should be run over at least a 2-week period. The short-term, carbon-14 stimulation-type assay that has been used[45] is not a reliable technique for measuring algal-available P because it does not provide the opportunity for bacteria in the samples to mineralize the readily mineralizable organic phosphorus compounds or phosphorus sorbed on organic detritus in the sample. When measuring algal-available P from wetland drainage, rather than the typical 2-week incubation period, a 1-month incubation at about 20°C in the dark is suggested to allow enough time for mineralization to take place. This extended bacterial mineralization period allows soluble, but unavailable, forms of P to be converted to available forms as they might eventually be transformed in a water body. This same approach should be used to investigate the availability of "unavailable" forms of phosphorus derived from the hydrolysis of condensed phosphates.

Stream Sampling

Some monitoring programs provide for monthly sampling of tributaries. This method can easily result in an incorrect assessment of algal-available or total P loads. It is particularly important in assessing algal-available P loads to sample during high flows, when the greatest transport of particulate P to a water body occurs. If possible, several special-purpose samplings should be made during the high flow periods. Guidance for sampling lakes, tributaries, sediments, and P sources and for the proper statistical analysis of sampling data is available elsewhere.[46] A 14- to 28-day interval is the minimum recommended for sampling tributaries to avoid risk of significant errors.[47] The actual sampling interval used should be determined by dividing the year into equal flow periods, so that all samples represent a more or less equal flow volume, rather than sampling at an arbitrary time interval. For "event response" streams in which phosphorus concentrations are higher during high-flow periods, 15 or 20 grab samples should be taken during each of two or three of the largest storms.[48] Generally, investigators have found that the continuous recording of stream flow gives much more reliable estimates of water inflow for computing P loads than periodic flow measurements.

Sampling frequency of point sources that have diurnal variations in loading also can be important in a field program. The effect of variations in point-source loading was measured at three stations downstream from the sewage treatment plant in Bucyrus, OH.[49] Samples were taken three times per day and 12 times per day. Mixing between the source and downstream sampling station tended to dampen wide diurnal fluctuations (Figure 2). The diurnal loading variation data were used to calibrate a phosphorus transport and deposition model for a section of the Sandusky River.

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[43]APHA.

[44]G. F. Lee, et al. (1980).

[45]R. L. Thomas, and Munawar, M., "The Delivery and Bioavailability of Particulate Bound Phosphorus in Canadian Rivers Tributary to the Great Lakes," *Proc. of the International Conference Management on Strategies for Phosphorus in the Environment* (Selper Ltd., London, 1985).

[46]K. H. Reckhow, and S. C. Chapra, *Engineering Approaches for Lake Management, Vols I and II* (Butterworth Publishers, 1983).

[47]K. H. Reckhow and S. C. Chapra.

[48]*Phosphorus Management of the Great Lakes*, Final Report of the Phosphorus Management Strategies Task Force (July 1980).

[49]D. B. Baker (1982).

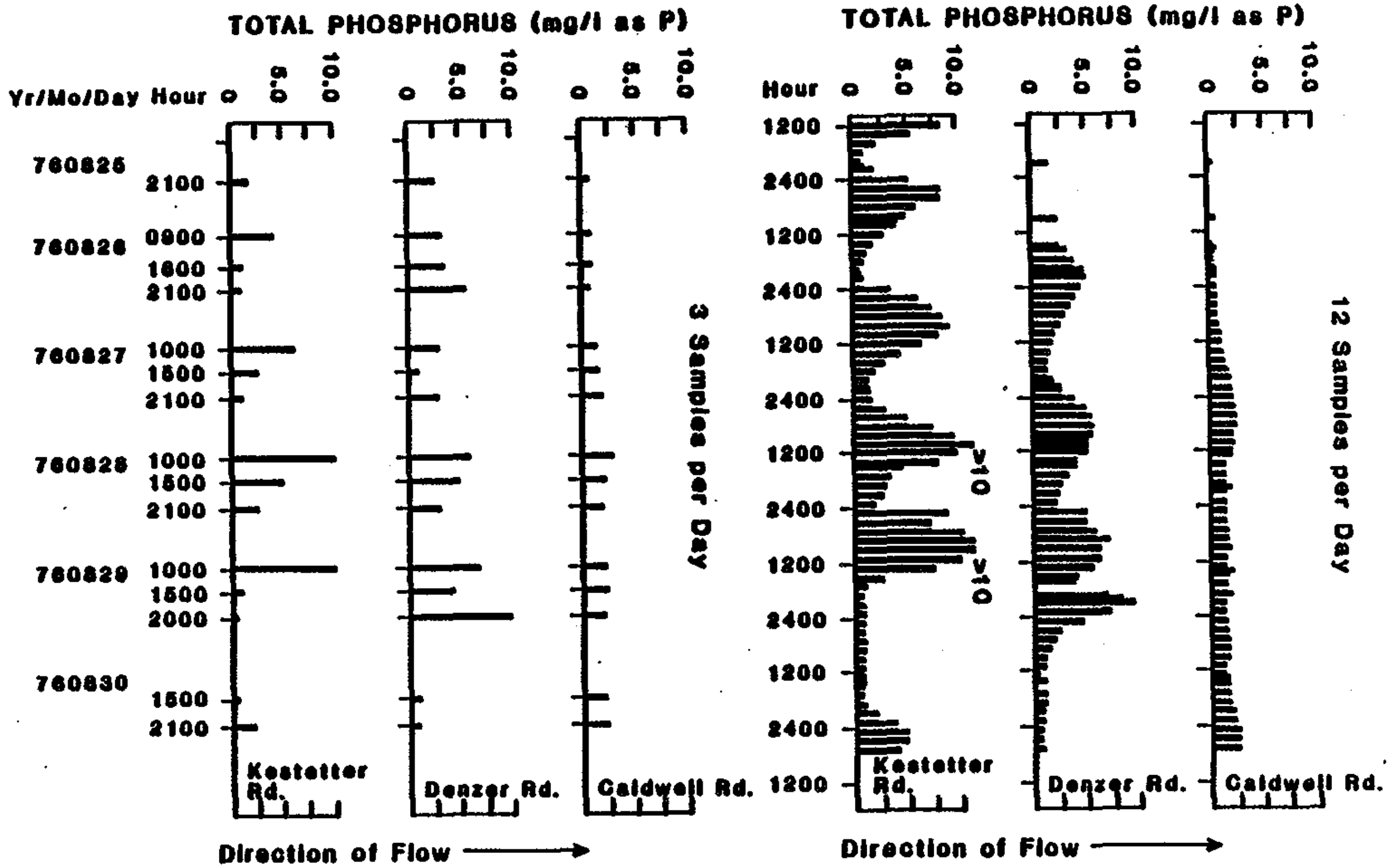

Figure 2. Phosphorus concentration profiles downstream from sewage treatment plants based on different sampling intervals. (Source: D. B. Baker, Fluvial Transport and Processing of Sediment and Nutrients in Large, Agricultural River Basins [U.S. Army Corps of Engineers, Buffalo District, February 1982].)

A field study will be conducted to determine the amount of available phosphorus from upstream sewage treatment plants and nonpoint sources in Arkansas and Oklahoma. This cooperative effort between the U.S. Environmental Protection Agency and other organizations is scheduled to begin in fall 1986, and will involve stream and runoff sampling and phosphorus loading calculations (J. Gakstatter, U.S. Environmental Protection Agency, [USEPA], Corvallis Environmental Research Lab, Corvallis, OR, unpublished results).

4 MODELING PHOSPHORUS LOAD-EUTROPHICATION RESPONSE RELATIONSHIPS

In developing a eutrophication management strategy, a key asset is being able to model the relationships between phosphorus load eutrophication-related water quality response. Once this information is established for a water body, it is possible to evaluate the impact of altering the phosphorus load on eutrophication-related water quality. Probably the most reliable eutrophication modeling approach is the Vollenweider-OECD method.

Vollenweider-OECD Approach: Summary

A wide variety of models is available, ranging from simple statistical correlations to highly complex deterministic models. Although it is relatively easy to adjust models to fit an observed load-response relationship for a particular water body, there is no assurance that the adjusted models will be able to predict the new degree of eutrophication resulting from a major change in the phosphorus load. However, the Vollenweider-OECD modeling relationship, based on P load-eutrophication response data from approximately 400 water bodies world wide, has predictive capability.[50]

The Vollenweider-OECD model is appropriate for Lake Champlain because it does not have to be calibrated to a specific water body and, given a phosphorus load, the change in euthrophication potential can be predicted. This predictive capability is useful in developing a phosphorus management strategy.

During the 1970s, the Organization for Economic Cooperation and Development (OECD) sponsored a 5-year study of the relationships between nutrient loading and eutrophication-related water quality response in water bodies in the United States, Canada, Australia, Japan, and 14 countries in Western Europe. A primary purpose of this study was to try to quantify, for a broad range of water body types, the relationship between the nutrient loading (especially phosphorus) and eutrophication-related water quality responses. About 34 water bodies or parts thereof in the United States that had already been studied intensively for nutrient loading and response characteristics were included in this study; new studies were initiated on the water bodies in other countries.

Qualitative Load-Response Relationship

Because more data were available on the U.S. waters, this group was the first to be evaluated using theoretical approaches developed by Vollenweider.[51] Vollenweider had begun to quantify relationships between nutrient loading and eutrophication-related water quality as assessed by "eutrophic," "mesotrophic," and "oligotrophic" designations

[50]R. A. Jones, and G. F. Lee, "Vollenweider-OECD Eutrophication Modeling Update," presented at the International Conference on Management Strategies for Phosphorus in the Environment, Lisbon, Portugal (July 1985).

[51]W. Rast and G. F. Lee (1978); G. F. Lee, R. A. Jones, and W. Rast, "Eutrophication of Water Bodies: Insights for an Age-Old Problem," *Environ. Sci & Technol.*, Vol 12 (1978b).

(defined in Table 1) based on about 20 (mostly European) water bodies.[52] One of the relating factors found was the water body's mean depth; the greater this value, the greater the phosphorus loading could be before "eutrophic" conditions would occur. This finding reflects the greater dilution capacity of deeper water bodies and proportionately smaller photic zones, allowing the deeper waters to have higher P loads without stimulating greater amounts of algae. It also reflects the presence of thermal stratification in deeper water bodies which limits nutrient recycling from the deeper waters during summer.

To account for the effects of fast or slow flushing rates of water through a water body on nutrient use by planktonic algae and for the impact of surface area on the water receiving light, Vollenweider modified the relationship between P load and mean depth to include hydraulic residence time and water body surface area. Figure 3 shows the result. When the phosphorus load per unit area of water is plotted as a function of the ratio of mean depth to hydraulic residence time, eutrophic water bodies tend to cluster in one area whereas oligotrophic waters tend to cluster in another. This early finding was substantiated when the U.S. OECD water bodies were plotted on this model, as shown in Figure 3.

It is most important to understand that the positioning of the "excessive" and "permissible" lines in Vollenweider's qualitative P loading evaluation is related to the earlier work by Sawyer in investigating public response to southern Wisconsin lakes in the mid-1940s.[53] These lines have no general significance in developing acceptable phosphorus loads for U.S. water bodies, much less those in other countries. What Figure 3 illustrates is a gradation or continuum of eutrophication-related water quality such that, for any given mean depth-to-hydraulic residence time ratio, the greater the P loading per unit area of water body, the more fertile the water tends to be.

Normalized P Loading-Planktonic Algal Chlorophyll

The overall relationship shown in Figure 3 does suggest a coupling of eutrophication-related water quality response with P loading, mean depth, and hydraulic residence time. The focus on P loading was based on several considerations—mainly that P most frequently controls algal growth and that P control is more readily effected, less costly, and easier to manage at the source than nitrogen. Therefore, to quantify these relationships, Vollenweider took his work a step further and plotted, for a group of water bodies, average planktonic algal chlorophyll concentration as a function of the annual P load normalized by the water's area, mean depth, and hydraulic residence; the log-transformed points appeared to form a linear relationship.[54] The same types of computations were made for the U.S. OECD water bodies and a line of best fit was developed which was similar to that developed by Vollenweider.[55] This line of best fit

[52] R. A. Vollenweider, *Scientific Fundamentals of the Eutrophication of Lakes and Flowing Waters, With Particular Reference to Nitrogen and Phosphorus as Factors in Eutrophication*, Technical Report DA 5/SCI/68.27 (Organization for Economic Cooperation and Development, 1968); R. A. Vollenweider, "Input-Output Models With Special Reference to the Phosphorus Loading Concept in Limnology," *Schweiz. A. Hydrol.*, Vol 37 (1976).

[53] W. Rast and G. F. Lee (1978).

[54] R. A. Vollenweider (1976).

[55] W. Rast and G. F. Lee (1978).

Table 1

Public's Perception of Change in Eutrophication-Related Water Quality*

Trophic State	Chlorophyll (μg/L)	Detectable Change in Annual Average Water Quality Based on Chlorophyll (μg/L)
Oligotrophic	<2	0.5
Mesotrophic	4 to 10	1 to 2
Hypereutrophic	45 (Pea Soup)	10

*Based on the Vollenweider—OECD P load-euthrophication relationships, a 20 percent change in total P load to a lake is needed to produce a change in water quality detectable by the public.

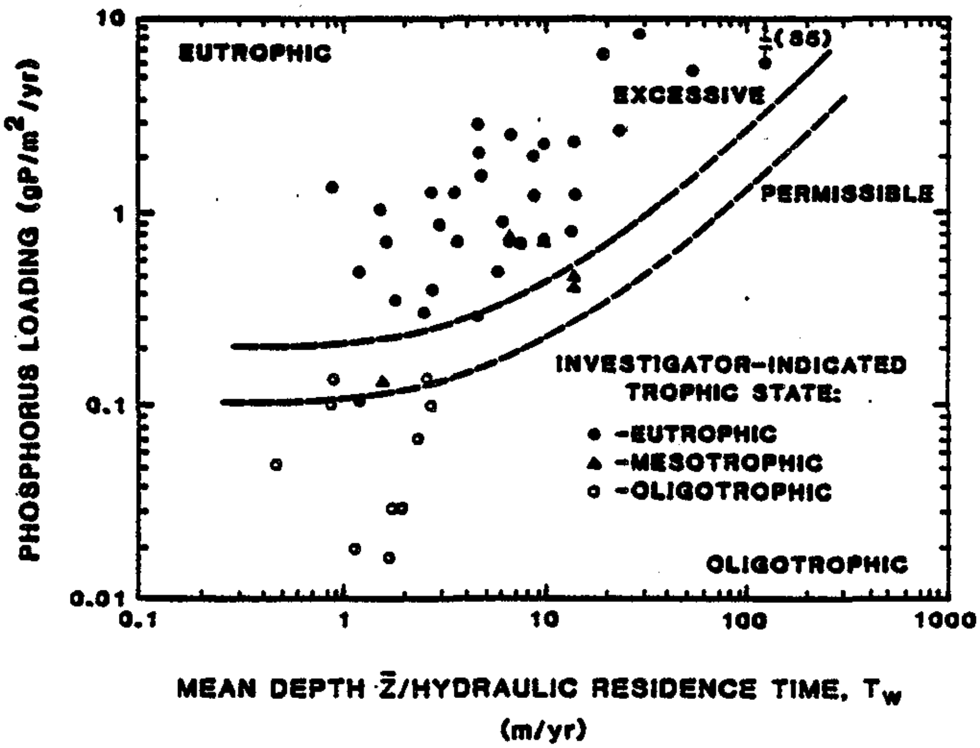

Figure 3. Vollenweider mean depth-hydraulic residence time phosphorus loading relationship for U.S. OECD lakes and reservoirs. (Source: W. Rast and G. F. Lee [1978]. Used with permission.)

was later updated to include about 40 additional U.S. water bodies for which data were available or accessible; this new line of best fit was essentially the same as the earlier one.[56]

Rast and Lee also had expanded the scope of the original normalized P loading-chlorophyll relationships by developing analogous relationships between normalized P loading and Secchi depth (water clarity), and between normalized P loading and hypolimnetic oxygen depletion rate based on U.S. OECD data.[57] These relationships were also updated later.[58] Figure 4 shows the three normalized P loading-eutrophication-related water quality response relationships.

Normalized P Load-Fish Yield

In addition to the relationships just described, work has been conducted relating eutrophication-related water quality with overall fish yield.[59] Figure 5 shows the relationship developed based on information from the literature indicating that the greater the normalized P load, and hence, the greater the algal biomass supported, the larger the fish yield. This plot also illustrates that the most desirable aesthetic quality—low productivity—and most productive fishery typically are antagonistic goals for water quality in a water body. If the yield of desirable fish were plotted as a function of normalized P load, however, there would likely be at least two breaks in the slope of the line of best fit in Figure 5 where the slope would decrease. One would correspond to the normalized P load that would cause enough algal growth to deplete oxygen in the hypolimnetic waters, thus precluding certain cold-water, desirable fish such as the salmonids. Also, in highly eutrophic water bodies, the fish tend to be considerably smaller and hence less desirable. The relationship shown in Figure 5 nonetheless provides a starting point for defining these relationships in more detail and will be especially useful in developing countries where the concern is for food, rather than esthetic enjoyment of water.

Demonstration of Predictive Capability

Perhaps the most important aspect of the Vollenweider-OECD eutrophication modeling approach is the demonstration and verification of its predictive potential. Although other statistical models are similar in principle, the Vollenweider-OECD load-response model has an extensive, varied, recently updated data base, adding confidence for its widespread applicability to many water body types.[60] The dynamic types of eutrophication models, and indeed other statistical load-response regressions, have not been demonstrated to be reliable in predicting new steadystate eutrophication-related water quality characteristics after substantial P loading changes. This step is crucial to the use of any modeling approach for estimating a nutrient control program's impact on water quality.

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[56] R. A. Jones, and G. F. Lee, "Recent Advances in Assessing the Impact of Phosphorus Loads in Eutrophication-Related Water Quality," *Journal Water Research*, Vol 16:(1982a), pp 503-515.
[57] W. Rast and G. F. Lee (1978).
[58] R. A. Jones and G. F. Lee (1982a).
[59] G. F. Lee and R. A. Jones, *Effect of Eutrophication on Fisheries*, Occasional Paper No. 74 (Department of Civil and Environmental Engineering, New Jersey Institute of Technology, 1985a).
[60] R. A. Jones and G. F. Lee (1985).

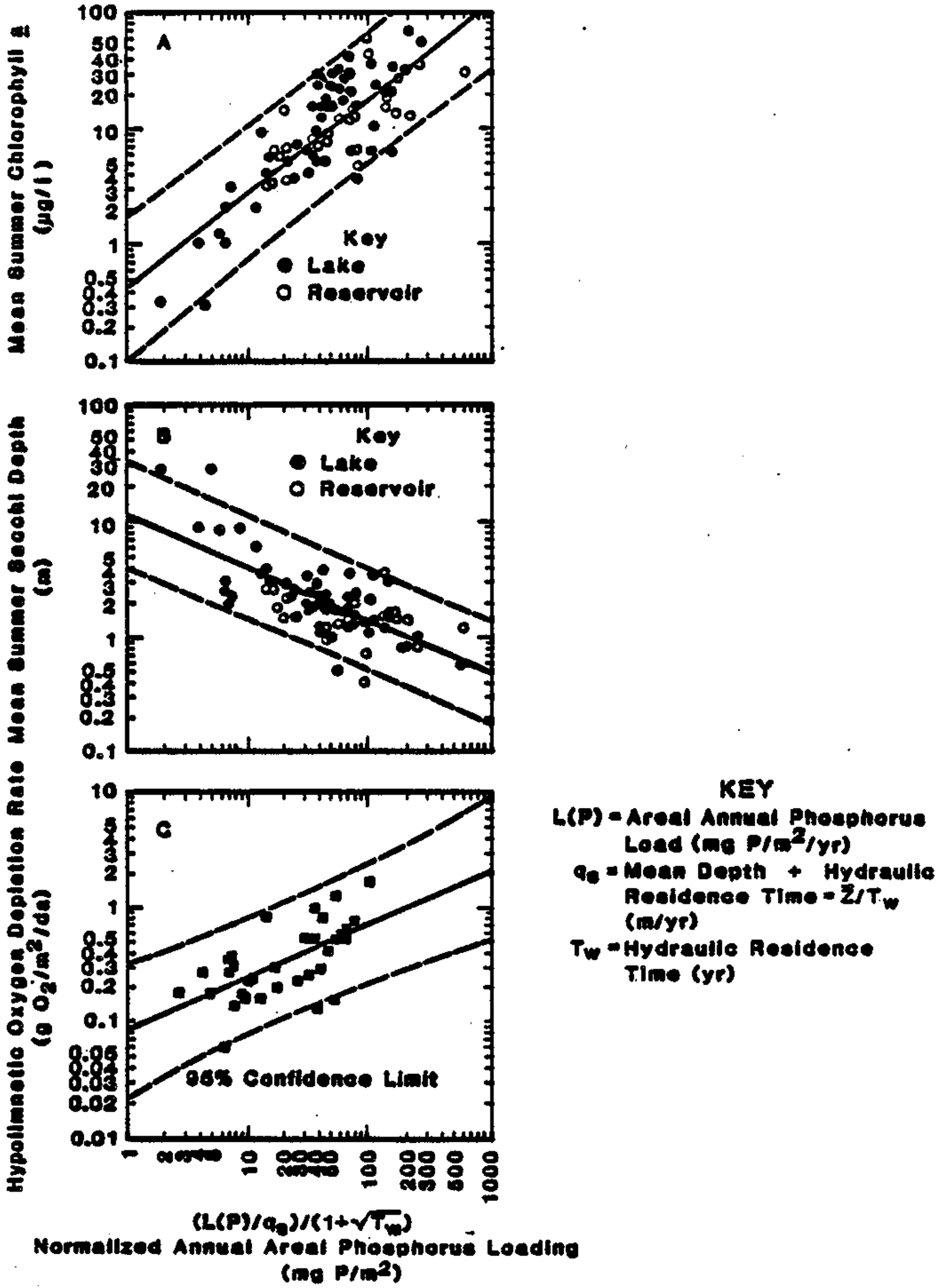

Figure 4. **Updated P load-eutrophication-related water quality response relationships for U.S. Water bodies.** (Reprinted with permission from J. Water Research, Vol 16, R. A. Jones and G. F. Lee, "Recent Advances in Assessing the Impact of Phosphorus Loads in Eutrophication-Related Water Quality," 1982, Pergamon Press, Ltd.)

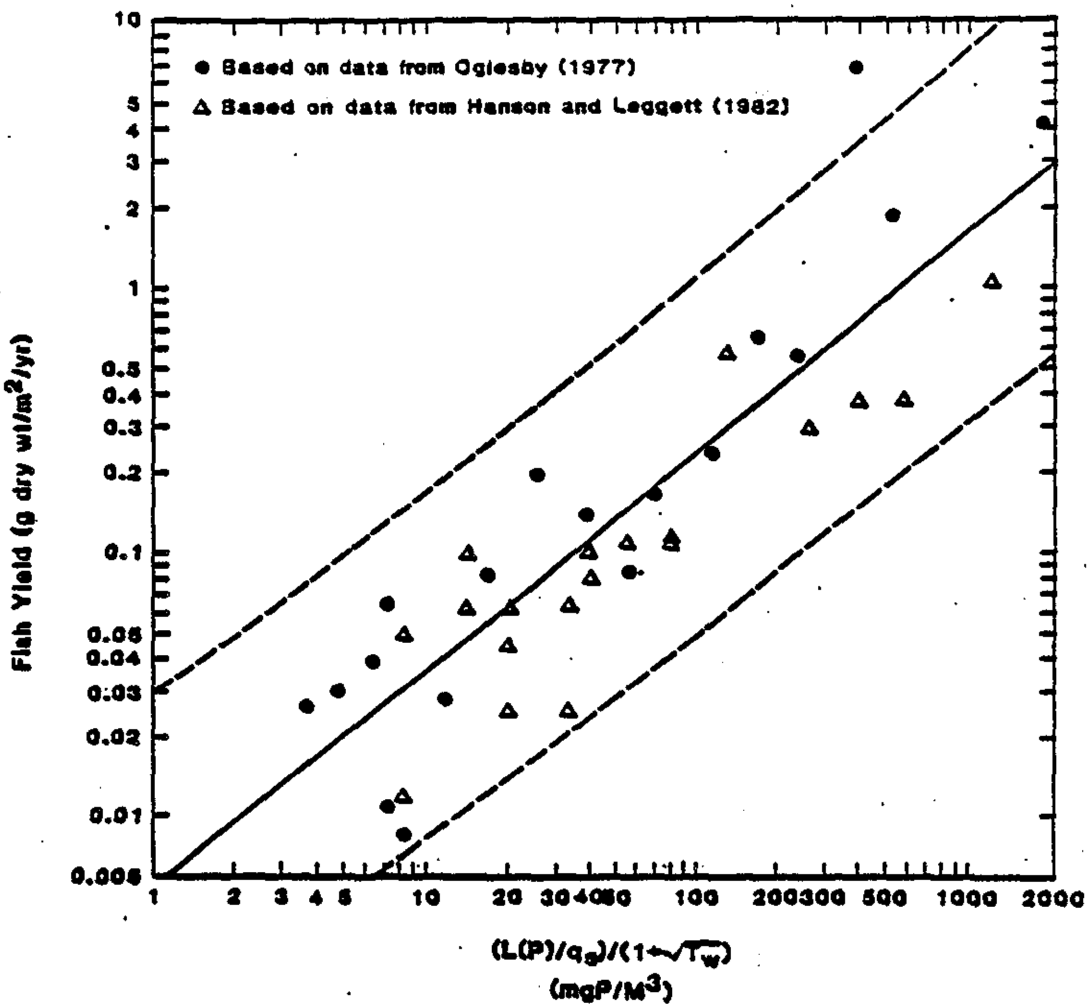

Figure 5. **Phosphorus load-fish yield relationship.** (Source: G. F. Lee and R. A. Jones [1985a]. Used with permission.)

To evaluate the predictive potential of the U.S. OECD normalized P loading-eutrophication-related water quality relationships, one study evaluated the data on nutrient load and water quality response for water bodies that had had P load reductions.[61] Although many eutrophication studies and nutrient load reduction projects for eutrophication control had been done, few studies had developed enough data to document quantitative changes in eutrophication-related water quality; the changes often were expressed qualitatively. Frequently, data had been collected a few weeks before enactment of a control measure; the parameters measured were not proper indications of "response"; or the "post control" monitoring was terminated before the water body had had a chance to equilibrate to its new loading conditions (about three times the P residence time, which often translates to a recovery period on the order of 1 to 3 years). The data available on load and response characteristics were collected and the model's predictive ability was evaluated for about a dozen water bodies. Figure 6 shows results of this evaluation for the chlorophyll response parameter.[62]

[61] W. Rast, R. A. Jones, and G. F. Lee, "Predictive Capability of U.S. OECD Phosphorus Loading-Eutrophication Response Models," *Journ. Water Pollut. Control Fed.*, Vol 55 (1983).

[62] W. Rast, et al. (1983).

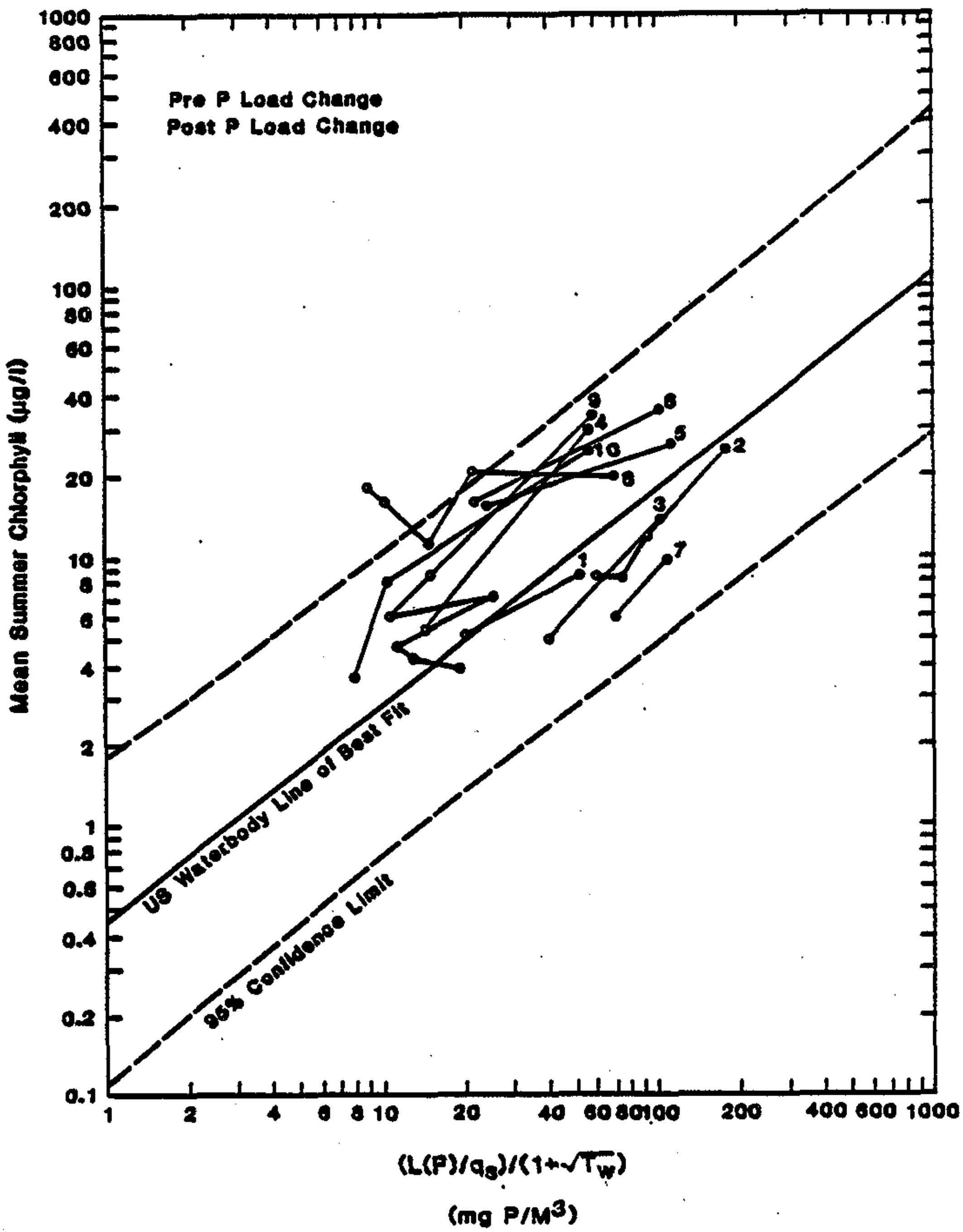

Figure 6. Phosphorus loading–chlorophyll couplings before and after P load changes. (Source: W. Rast, et al. [1983]. Used with permission.)

Figure 6 shows that, for any given water body, the load-response couplings track more or less parallel to the U.S. waters' line of best fit.[63] Careful comparison of the measured chlorophyll values after load reduction with those predicted based on the U.S. line of best fit shows an error typically within a few percent of the analytical error expected for chlorophyll analyses. More importantly, this examination also shows that the differences between measured and predicted chlorophyll are not great enough to trigger a different decision regarding treatment. This finding is significant, especially since the literature gave generally too little information to screen the data for consistency, and to ensure that all before and after load change data were representative of conditions present, that the data represented equilibrium conditions, and that no other perturbation was occurring to alter the load-response relationships. More detailed discussions of the approach's predictive ability and how it can be used in water quality management are available elsewhere.[64]

Updated Load-Response Relationship

Work in the United States and other countries in exploring the Vollenweider-OECD model's applicability has continued beyond the original work to establish load-response relationships. This approach has been applied to approximately 50 additional water bodies in the United States and to about 70 in other countries.[65] The load-response relationships for these water bodies have been in keeping with those developed on the basis of OECD[66] and U.S. OECD[67] data. Since the models originally were developed on the basis of water bodies having a certain range of morphological and hydrological characteristics, much of the recent work has focused on waters with properties different from those of water bodies in the original OECD study. Of special interest has been waters that seemingly did not fit the U.S. water body load-response relationships. Overall, the data available to date covers a broad spectrum of water bodies, from small lakes of 1 ha to large water bodies such as Lake Superior with a surface area of 8 million ha. The range includes shallow (1.5 m) water bodies to those having a maximum depth of about 500 m (Lake Superior), with a mean depth of about 150 m (Lake Mjosa); the mean chlorophyll concentration ranges from more than 100 μg/L (hypereutrophic Fossil Creek Reservoir) to 0.3 μg/L (Lake Tahoe) and 0.12 μg/L (Lake Vanda). Lakes, reservoirs, and estuarine systems all behave the same in planktonic algal response to nutrient loading. Those in warm climates such as Texas and Florida in the United States and in Tunisia and those in the cold Nordic and Antarctic climates are similar in use of phosphorus for planktonic algae growth. Water bodies in the Northern Hemisphere countries have the same load-response relationships as those in the Southern Hemisphere. For no other modeling approach has this broad a spectrum of water body types been included.

Figure 7 shows the locations of all water bodies evaluated to date--the U.S. OECD, international OECD, and additional U.S. and non-U.S. systems studied by the authors--on the Vollenweider-OECD normalized P load-chlorophyll plot. Comparing this figure with

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[63] W. Rast, et al. (1983).

[64] W. Rast, et al. (1983); G. F. Lee and R. A. Jones, "Evaluation of the Effectiveness of Phosphorus Removal from Domestic Wastewaters on the Eutrophication of Receiving Waters," Presented at the International Conference on Management Strategies for Phosphorus in the Environment, Lisbon, Portugal (1985b).

[65] R. A. Jones and G. F. Lee (1985).

[66] Organization for Economic Cooperation and Development (OECD), *Eutrophication of Waters, Monitoring, Assessment, and Control* (Selper Ltd., London, 1985), pp. 365-370.

[67] R. A. Jones and G. F. Lee (1985).

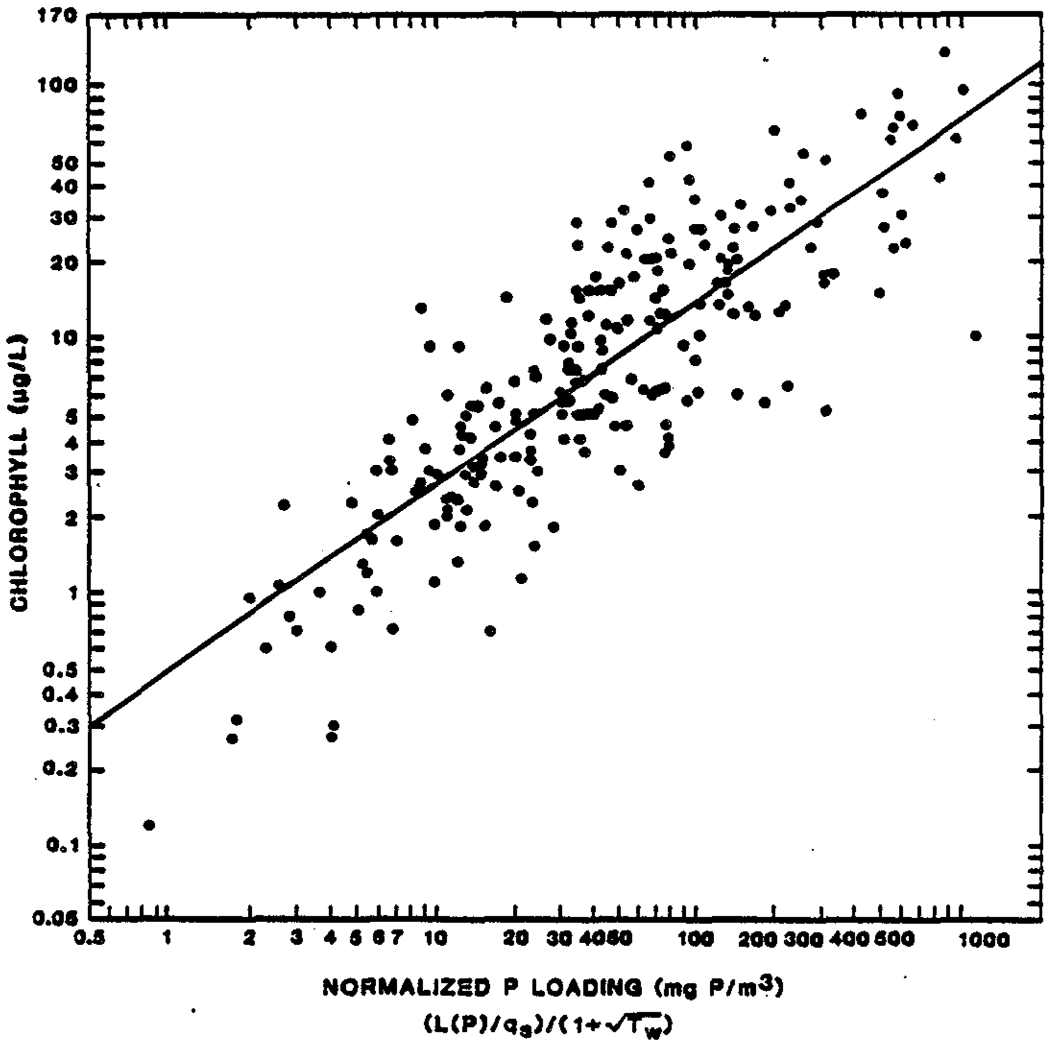

Figure 7. Updated OECD eutrophication study phosphorus load-chlorophyll relationship. (Source: R. A. Jones and W. F. Lee [1985]. Used with permission.)

graph A of Figure 4 shows that the lines of best fit are essentially the same. The fact that more than 325 data points describe the same relationship as described previously with the U.S. data, and the fact that these waters represent such a wide range in morphological, hydrological, biological, and chemical characteristics, demonstrate the universality of this relationship. These facts provide internal verification of this model's ability to predict changes in eutrophication-related water quality response (as measured by chlorophyll) that will result from changes in the normalized P loading. It is clear that, independent of about every other parameter, if the phosphorus load to any water body is normalized by the water body's mean depth, hydraulic residence time, and surface area in accordance with the Vollenweider model and an initial load-response coupling is known, the planktonic algal chlorophyll concentration can be estimated reliably for the new loading. This ability, coupled with the before and after P load change predictions, gives a considerable level of confidence in using this approach to estimate changes in eutrophication-related water quality.

Relationship Between Normalized P Loading and In-Lake P Concentration

Vollenweider originally developed the normalized P loading term by relating the water body's P load to its in-lake phosphorus concentration. This in-lake phosphorus concentration can be directly correlated with the amount of planktonic algae supported based on the relatively constant algal stoichiometery of 106 carbon to 16 nitrogen to one phosphorus atom. Although the Vollenweider-OECD load-response model uses total P loading, the load is normalized to account for the fact that not all total P entering a water body stays in the water column to support planktonic algal growth. Actual units of the normalized P loading term are hybrid, but the term itself represents the average in-lake P concentration for water bodies with typical amounts of their P load in both available and unavailable forms. In actuality, this is not a one-to-one relationship because of differences in the P's algal availability. For the OECD water bodies, about 80 percent of the P load appears to be available to support algal growth. The normalized P loading term, however, can be related empirically to the average in-lake P concentration. Rast et al.[68] found the following relationship for the U.S. water bodies evaluated by Jones and Lee:[69]

$$[P]_{\text{in-lake}} = (\text{Normalized P loading})^{0.81} \qquad \text{[Eq 1]}$$

where $P_{\text{in-lake}}$ = the average annual in-lake phosphorus concentration; normalized load = $[L(P)/q_s]/(1+\sqrt{T_w})$ where L(P) = phosphorus loading in mg/m^2 lake-yr, q_s = corrected value for flow in m/yr, and T_w = hydraulic resistance time in yr; and units for P are mg/m^3. This relationship is essentially the same as that found by Vollenweider for the overall OECD water body data base.

Use of this equation for determining the P concentration from the loading term assumes that the water body will have an available phosphorus load around 80 percent of the total phosphorus load; the water bodies on which Equation 1 was developed do not have heavy loads of particulate phosphorus. Therefore, applying this equation to a water body with much different P load characteristics could result in a significantly different relationship from that shown in Equation 1.

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[68] W. Rast, et al. (1983).
[69] R. A. Jones and G. F. Lee (1982a).

Other Modeling Studies

The U.S. Army Corps of Engineers has used and modified mechanistic and empirical models for predicting in-lake concentrations of phosphorus and nitrogen species, chlorophyll-a and Secchi depth.[70] The model inputs were: influx of phosphorus and nitrogen species, reservoir mean depth, and hydraulic residence time. Bioavailable P entering a reservoir was considered to be influent ortho P plus a fraction of the influent non-ortho P. Empirical models developed for predicting in-lake P concentrations had weighting factors for "direct point source," tributary ortho P, and tributary non-ortho P entering a reservoir. It was found that, in the modeling scheme with the smallest error, tributary ortho P was significant compared to other P sources entering the reservoir. The reservoirs in the data base used have most influent P derived from nonpoint sources. Other dynamic models have been used to evaluate eutrophication control measures for phosphorus from point and nonpoint sources. A study on the transport and transformation of phosphorus in Lake Erie tributaries[71] found that, downstream from sewage treatment plant discharges, the total phosphorus and orthophosphate were quickly adsorbed onto river sediments or microorganisms. A dynamic phosphorus transport model was used to calculate phosphorus deposition and resuspension patterns under various flow conditions (Figure 8). According to the model, total phosphorus and orthophosphate are deposited during low flow; during high-flow periods, only total phosphorus is resuspended. The orthophosphate probably is transported downstream.

That study calculated the cumulative probability that phosphorus from a given point source would be deposited before traveling a stated number of miles. Figure 9 shows cumulative probabilities for five point sources under particular flow conditions. Given the flows' histogram, the residence time in the river basin was calculated. There is no indication that this study has influenced eutrophication management policy in the basin.

The Hydrologic Simulation Program, FORTRAN version (HSPF) model was developed by the USEPA for projecting water quality, given phosphorus loading changes and eutrophication management options.[72] HSPF consists of three submodels: Hydrologic Submodel, which calculated evaporation, runoff, soil moisture retention, and stream flow, given rainfall, soil type, and other factors; Nonpoint Pollution Submodel, which calculates runoff and erosion, given storms, land use, tillage practices, and output from the first submodel; and the Receiving Water Submodel, which combines stream flow and pollutant loadings from the first two submodels and predicts daily pollutant concentrations and transport at various points in the watershed. Estimates are made hourly for organic phosphorus sedimentation, conversion between organic/inorganic forms, phytoplankton growth, and sediment release of phosphorus.[73] A version of this model was used to predict the impact of nonpoint source controls compared to point source controls in the Occoquan watershed, which feeds the Occoquan Reservoir (unpublished results). By testing alternative land-use plans using the Occoquan River Basin Computer Model, it was determined that controlling urban nonpoint source loads

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[70]W.W. Walker *Empirical Methods for Predicting Eutrophication in Impoundments* TR-E-81-9-3 (U.S. Army WES, March 1983).

[71]U.S. Army Corps of Engineers, *Lake Erie Wastewater Management Study Methodology Report* (USACE Buffalo District, 1979).

[72]U.S. Environmental Protection Agency, *Application Guide for HSPF*, EPA 600/3-84-065 (1984).

[73]J. P. Hartigan, et al., "Post-Audit of Lake Model Used for NPS Management," *Journal of Environmental Engineering, ASCE*, Vol 109, No. 6 (December 1983).

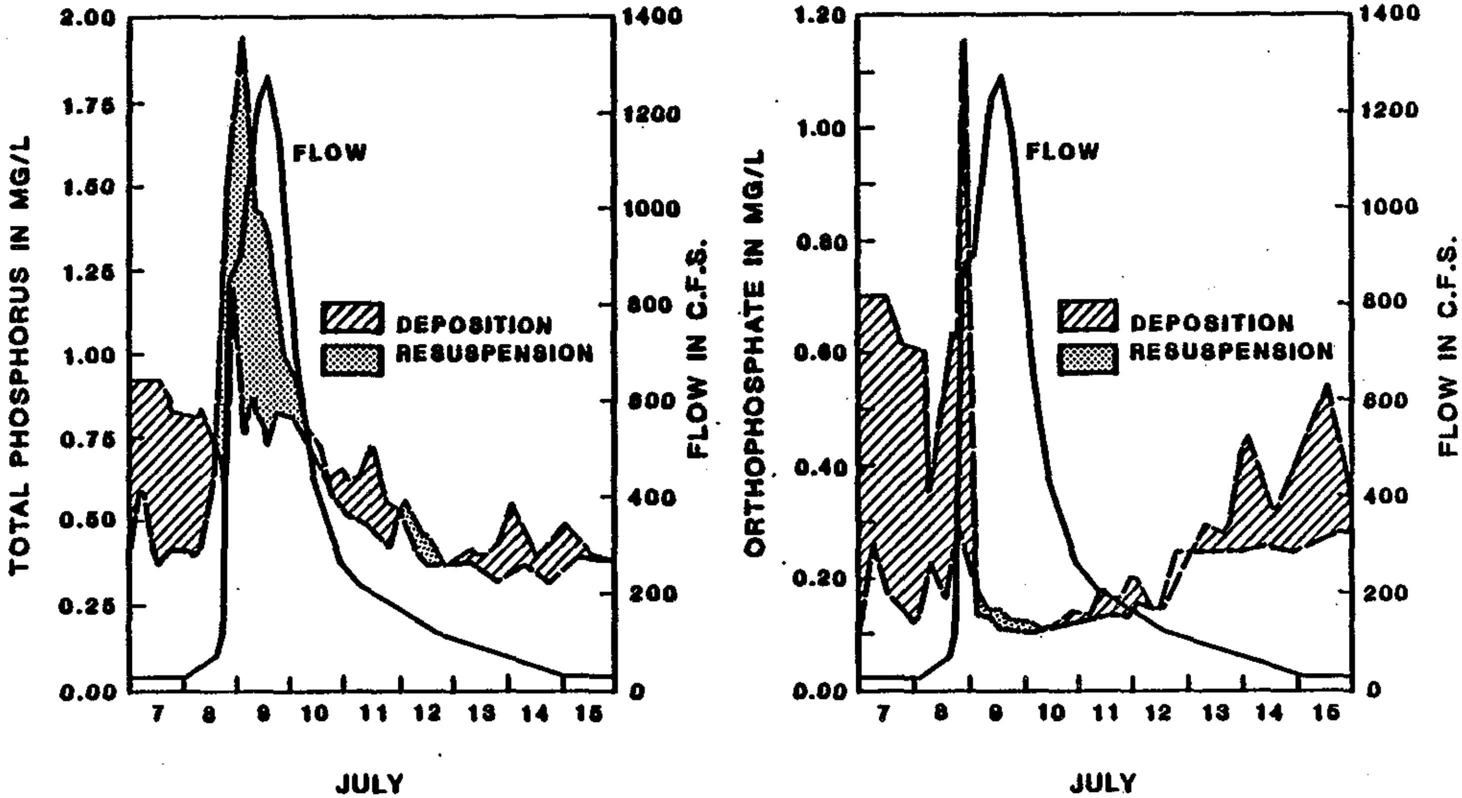

Figure 8. Deposition and resuspension of (a) total phosphorus and (b) orthophosphate in the Sandusky River near Upper Sandusky—storm beginning 7 July 1976. (Source: U.S. Army Corps of Engineers, <u>Lake Erie Wastewater Management Study</u>, Methodology Report [March 1979].)

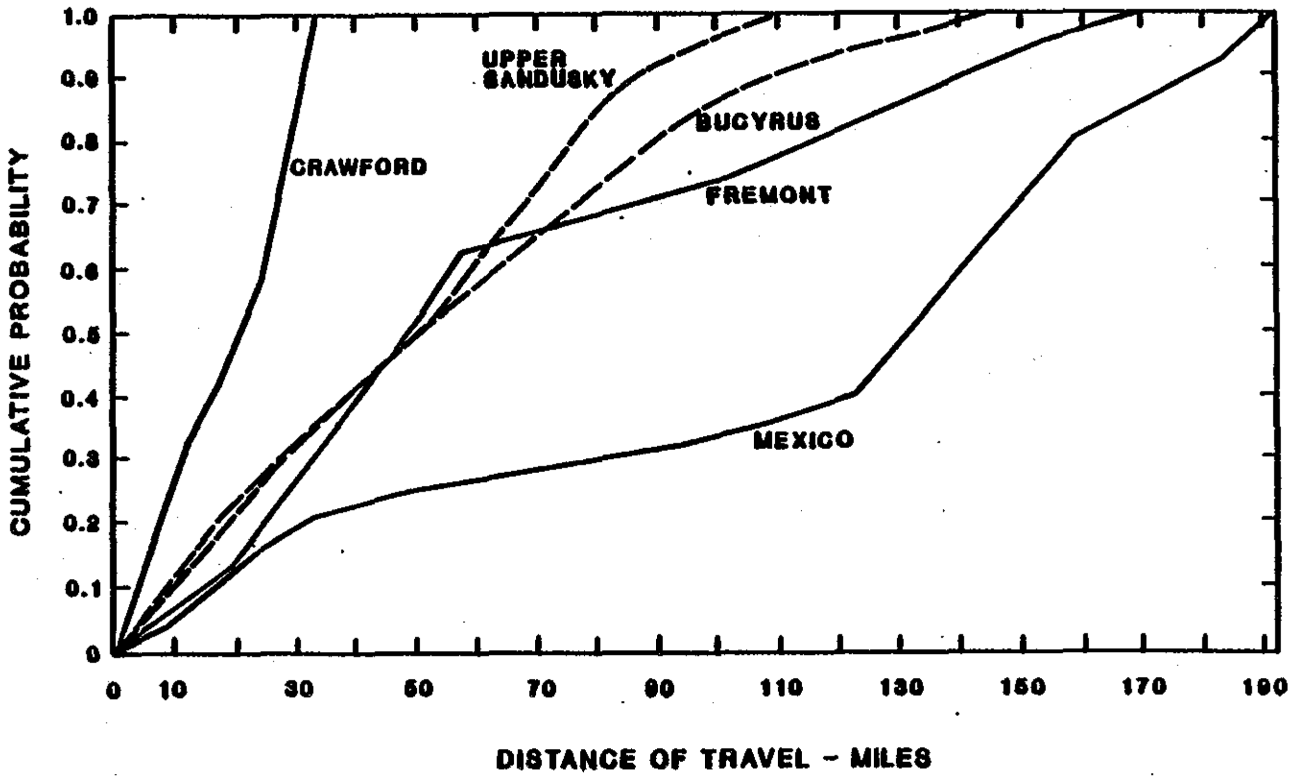

Figure 9. **Cumulative probability that total phosphorus will be deposited in-stream before traveling the stated distance—Sandusky River basin, 12 July 1974. (Source: U.S. Army Corps of Engineers, Lake Erie Wastewater Management Study, Methodology Report [March 1979].)**

could complement phosphorus removal by sewage treatment plants.[74] Chapter 5's discussion of optimal control measure combination describes additional model studies.

Estimating Nutrient Loads

To estimate the impact of altering the phosphorus load on a water body's eutrophication-related quality, it is necessary to estimate the system's total phosphorus load. To do this involves measuring the phosphorus load that enters the water body via tributaries and other point sources. Water samples are collected at approximately weekly intervals over at least 1 year and analyzed for total P. At the point of sampling, the river or source is also gauged to measure the total water flow. The total amount of added phosphorus can then be estimated. (Other guidelines are in Chapter 3.)

Models have been developed and calibrated for individual watersheds to calculate phosphorus loading to a water body. The Nonpoint Pollution Loading Submodel of the HSPF model calculated erosion and runoff water quality, given rainfall events and land-use characteristics. A sediment and nutrient transport model has been developed for the watershed at White Clay Lake, WI, to evaluate the effectiveness of sediment and nutrient control practices.[75] The model quantifies phosphorus reaching stream channels from manure stored in barnyards and spread on frozen ground. It also evaluates sediment-derived phosphorus and soluble reactive phosphorus loading based on average annual precipitation, land use, slope, soil type, plowing methods, and residue handling. After comparing monitored and predicted total P values for 6 years, it was concluded that the model predicts long-term averages reasonably well.

Estimates of phosphorus loads can be based on nutrient export coefficients and land use within the watershed. This approach evolved from the U.S. OECD eutrophication studies.[76] Table 2 summarizes nutrient export coefficients used in estimating the phosphorus loads to a water body and the impact of altering these loads on eutrophication-related water quality.

Nutrient export coefficients, developed in the mid-1970s, have been applied to a wide variety of situations in the United States and other countries and have been found to give reliable estimates for nutrient loads to water bodies. Rast and Lee have summarized the approaches for management strategies for water bodies.[77] This work also discusses various techniques found to be useful in evaluating the reliability of nutrient load estimates for a particular water body.

Public's Perception of Water Quality to Changes in Phosphorus Removal

In developing a phosphorus management strategy, it is important to consider the sensitivity of eutrophication-related water quality changes, as well as the public's perception of changes, with changes in phosphorus loads. Lee and Jones have discussed the public's ability to discern changes in eutrophication-related water quality at various trophic levels.[78] Table 1 (page 24) summarizes these results.

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[74] J. P. Hartigan, et al.

[75] L. A. Persson, et al.

[76] W. Rast and G. F. Lee (1978); G. F. Lee, et al. (1978).

[77] W. Rast and G. F. Lee, "Nutrient Load Estimates for Lakes," *J. Env. Eng. Div. ASCE*, Vol 109 (1983).

[78] G. F. Lee and R. A. Jones (1985b).

Table 2

Phosphorus Export Coefficients*

Land Use — Source	Total P Export Coefficient
Urban	0.1 $g/m^2/yr$**
Rural/agricultural	0.05 $g/m^2/yr$**
Forest	0.005 $g/m^2/yr$**
Rainfall/dry fallout	0.02 $g/m^2/yr$***
Domestic wastewater	1 kg/person/yr

*Source: W. Rast and G. F. Lee, "Nutrient Loading Estimates for Lakes," *J. Env. Eng. Div. ASCE*, Vol 109 (1983). Used with permission.
**Based on watershed area.
***Based on water body area.

As Table 1 indicates, some persons probably will be able to detect a 0.5-µg/L change in planktonic algal chlorophyll when the average chlorophyll concentration is on the order of 1 to 2 µg/L. Actually, the public may not see the change in green color (chlorophyll) but will see an increase or decrease in water clarity (Secchi depth) related to planktonic algal growth. Figure 10 shows the relationship between water bodies' planktonic algal chlorophyll concentrations and Secchi depths.[79] It is based on data in the literature for several hundred water bodies, excluding those with high levels of inorganic turbidity or color. As this figure shows for oligotrophic waters (those with planktonic algal chlorophyll concentrations of less than about 2 µg/L), relatively small changes in algal biomass chlorophyll result in major changes in water clarity (Secchi depth).[80] It has been estimated that some persons may be able to see a change in planktonic algal chlorophyll on the order of 1 to 2 µg/L in a mesotrophic water body (chlorophyll concentrations on the order of 5 to 10 µg/L); the public probably will be able to detect a 10-µg/L chlorophyll change in hypereutrophic waters.[81]

Based on the relationship in Table 1 and the Vollenweider-OECD eutrophication study results,[82] a 20 percent change in normalized phosphorus loading to a water body must occur before some persons would be able to discern a change in planktonic algal

[79]W. Rast and G. F. Lee (1978).
[80]G. F. Lee, R. A. Jones, and W. Rast, *Secchi Depth as a Water Quality Parameter*, Occasional Paper No. 101 (Department of Civil and Environmental Engineering, New Jersey Institute of Technology, submitted for publication).
[81]G. F. Lee and R. A. Jones (1985b).
[82]G. F. Lee and R. A. Jones (1985b); R. A. Jones and G. F. Lee (1982a).

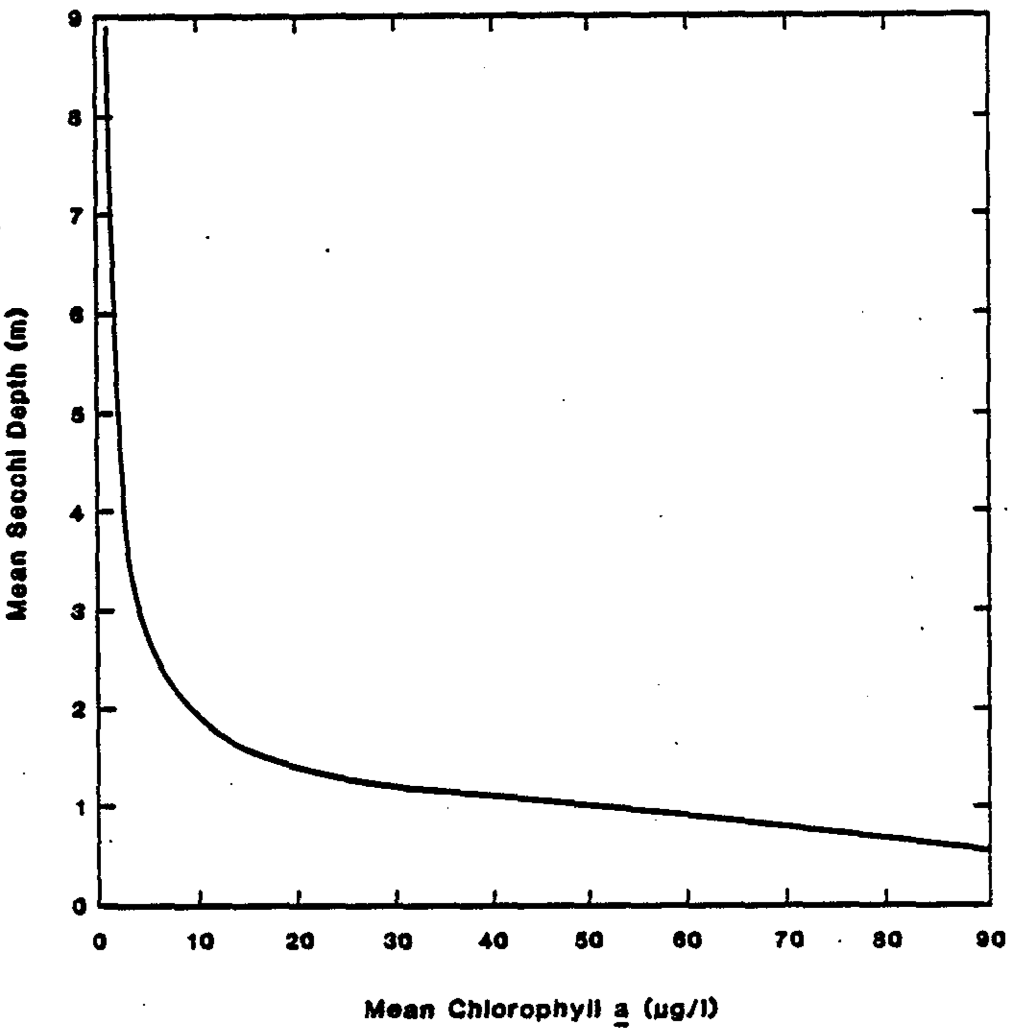

Figure 10. Secchi depth and chlorophyll-a relationship in natural waters (linear scale). (Source: W. Rast and G. F. Lee [1978]. Used with permission.)

chlorophyll arising from the change in phosphorus load. The percentage change that must occur in P load is independent of the water body's trophic state--a somewhat surprising result. However, this finding has been verified by workers in the eutrophication management field.

This 20 percent load change figure has important implications in developing a eutrophication management strategy for a water body; it means that changes in phosphorus load of less than this proportion will not have a discernible effect on eutrophication-related water quality as manifested by planktonic algal chlorophyll. For a P load change to produce a water quality change noticeable to most of the public, the total P load (which must be mostly available P) must change more than 30 percent; even changes on the order of 20 to 30 percent will not produce highly visible changes in water quality.

In the case of Vermont, this rationale could justify not requiring small communities to remove phosphorus from the domestic wastewaters, since the sum of all of these communities' phosphorus inputs will have a nondiscernible impact on Lake Champlain's water quality. Analysis of these inputs shows that about 32 communities have average wastewater flows of less than 0.5 mgd.[83] The estimated annual per capita cost for phosphorus removal in communities of this size or smaller is more than $200/year. The total estimated phosphorus load from all these communities is about 15 percent of the total P load to Lake Champlain from Vermont domestic wastewaters, which represents some 5 percent of the total P load to the lake. Because P load from these small plants is so little, waiving the requirements of phosphorus removal from these communities' domestic wastewaters will have no visible effect on Lake Champlain's overall water quality. However, as discussed earlier in this report, evaluation should be made to determine if phosphorus removal from a particular community's wastewaters will have important impact on nearshore water quality for a certain part of the lake.

The Vollenweider-OECD eutrophication modeling results can predict changes in planktonic algal chlorophyll, phytoplankton-related Secchi depth, and hypolimnetic oxygen depletion rates that will result from altering phosphorus loads. They cannot predict the impact of altering phosphorus loads on attached algae or macrophytes. It is likely, however, that changes in P load somewhat greater than 20 percent will be needed to cause major impact on the biomass of these aquatic plant forms because of (1) the higher rates of nutrient recycling in shallow waters and (2) the fact that attached macrophytes can use phosphorus from the sediments for growth. Therefore, phosphorus control programs should be directed toward managing phosphorus from sources that represent at least 20 percent and preferably greater than 30 percent of the total P load to a water body.

[83]E. Smelzer, "Phosphorus Removal Policy," Memorandum to R. Czaplinski, State of Vermont, Agency of Environmental Conservation, Department of Water Resources and Environmental Engineering, Montpelier, VT (21 September 1983).

5 ESTABLISHING PRIORITIES FOR PHOSPHORUS CONTROL

Approaches are presented for establishing phosphorus control programs that apply to various sources of phosphorus in Lake Champlain. Some approaches to formulating control programs in other drainage basins are included.

Domestic Wastewaters

An area of particular interest to Vermont is which approach should be adopted for prioritizing phosphorus removal at domestic wastewater treatment plants. Several years ago, the state adopted requirements for phosphorus removal to 1mg/L effluent P for all domestic wastewater treatment plants that discharge directly to Lake Champlain. This limit was recognized as technically and economically achievable. It exempted, however, all off-lake domestic wastewater treatment plants from this requirement.

The approach in Vermont differs from that adopted in the Great Lakes basin, where both on-lake and off-lake domestic wastewater discharges were considered equally important in the lakes' eutrophication. No consideration was given to loss of P by transport or attenuation. Further, states in the Great Lakes basin exempted very small communities from having to remove phosphorus from their domestic wastewaters, whether the treatment plants discharged directly to one of the lakes or to a tributary. This exemption was justified because the per capita cost for phosphorus removal increases dramatically with decreasing size of the treatment plants. In addition, in terms of overall P loading to a lake, the small treatment plants contributed a relatively minor amount of phosphorus. In contrast, Vermont requires that a much higher per capita cost for phosphorus removal be borne by individuals living in small communities for which the wastewater treatment plants discharge directly to Lake Champlain. Site-specific studies probably would show that these discharges have little or no impact on the eutrophication-related water quality within the lake or its nearshore water.

Although states in the Great Lakes basin require all medium to large communities to remove phosphorus to a relatively easily attainable effluent limit of 1mg/L total P, this level is achieved at a per capita cost of less than 1 cent per person. The cost in Vermont for very small treatment plants can be on the order of dollars per capita per day. At the same time, the state is allowing the discharge of large amounts of available phosphorus to river tributaries of Lake Champlain—some of which will reach the lake in a highly available form or will be readily converted to such a form within the lake. It is clear that Vermont needs to critically examine its phosphorus management strategy for Lake Champlain as well as other water bodies in the state to ensure the most technically valid, cost-effective eutrophication management.

It has been estimated that a P loading reduction of 163,000 kg/yr would be achieved by having the 20 largest Vermont sewage treatment facilities remove phosphorus to a level of 1 mg/L P. This level is approximately a two-thirds reduction in loading compared to the P loading without removal. This figure already takes into account a reduction due to a detergent P ban.

Henson [84] calculated the P loading from each Lake Champlain subbasin and the reduction in loading needed to reverse eutrophication in the bays. The Vollenweider model was used to determine the impact of loading changes in and target loads. In two subbasins (Burlington-Winooski and Otter Creek), implementation of both a P detergent ban and P removal at sewage treatment plants would not be enough to reduce the loading to achieve desired water quality conditions in the bays.

It is recommended that Vermont adopt a uniform 1 mg/L total P effluent standard for all domestic wastewater treatment plants. As part of adopting this standard, the state also should approve a variance procedure that would allow a community to demonstrate that 1 mg/L effluent P in its domestic wastewaters would not greatly impact the receiving water's eutrophication-related quality and that adoption of the requirement would represent an unreasonable financial burden on the community in cost per capita per day. This step would allow very small communities to become exempt from the requirement when their phosphorus loads have a relatively minor effect on both receiving and nearshore water quality. Because phosphorus is so important in causing water quality deterioration in Vermont's lakes, it seems reasonable that residents could afford a few cents per day for phosphorus control; dollars per day appears excessive. The exact per capita cost considered bearable, however, is a social issue that Vermont's residents must decide.

It is always important in making these decisions to remember the purpose of phosphorus removal—eutrophication management. In the mid-1970s, when Vermont first adopted phosphorus control legislation, information was just beginning to be developed on approaches for assessing the benefits of phosphorus control at a certain level. Today, by using the Vollenweider-OECD eutrophication study results, it is possible to predict reliably, for most water bodies, the impact of altering phosphorus loads a certain amount on the eutrophication-related water quality. As noted previously, a recent publication devoted to managing phosphorus in the environment describes the approaches to use for this purpose.[85]

In analyzing the benefits of controlling a particular phosphorus source to a certain degree, it is important to assess how much the phosphorus source influences the degree of eutrophication—not only for the water body as a whole, but also for the nearshore, high-use areas. Situations can develop in which relatively small domestic wastewater treatment plants' phosphorus discharges to a water body will have limited impact on overall water quality, but will have a major effect on an important beach, swimming area, or water-skiing section of the lake. For this reason, it is important that evaluations be site-specific, focusing on how all related waters are used. Although swimming, boating, and water-skiing opportunities are generally impaired as the water body becomes more fertile, another important use to consider is recreational fishing. There is a direct positive correlation between phosphorus loads to a water body and fish production in those waters.[86] It is therefore important to balance the conflicting uses of water bodies as part of developing a phosphorus management strategy.

For Lake Champlain and most other water bodies, the highest priority for phosphorus control should be given to large domestic wastewater treatment plants discharging directly into the lake or into one of the tributaries within a few miles of the

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[84]E.B. Henson, et al., *The Trophic Status and Phosphorus Loadings of Lake Champlain*, EPA 1600/3-77/106 (September 1976).
[85]G. F. Lee and R. A. Jones (1985b).
[86]G. F. Lee and R. A. Jones (1985a).

lake. Rarely would a variance from the 1-mg/L effluent P limitation be appropriate for such treatment plants. The second highest priority should be given to large domestic wastewater treatment plants discharging to the lake's river tributaries. The current exemptions from the 1 mg/L P domestic wastewater effluent standard by Vermont should be rescinded immediately for large communities such as Montpelier, Rutland, Barre City, Middlebury, Northfield, and Brandon. It is recommended that the Great Lakes basing philosophy of requiring phosphorus removal to 1 mg/L effluent P be adopted uniformly across the state of Vermont for all domestic wastewater treatment works when the cost to community residents is equal to or less than a few tens of cents per person per day. This policy would be valid unless the community could demonstrate that its wastewater effluent phosphorus does not enter a water body within the state or that at least 90 percent of the phosphorus discharged reaches the water body in an unavailable form.

The burden of proof in any disputed situation should rest with the community, not the state. Vermont's pollution control agency, like those nationwide, is inadequately funded to develop the most technically valid, cost-effective programs for managing the state water resources' quality. If communities or dischargers seeking variances must absorb the financial burden of the evaluation, the system would ensure payment by the persons who will gain the most by receiving an exemption. It is important that the issuing of such variances be as apolitical as possible, with primary considerations given to technical facts gathered from a site-specific evaluation of a particular discharge and the behavior of the phosphorus from this discharge within the receiving waters. It is important that the Department of Water Resources and Environmental Engineering establish well defined guidelines on how a community should proceed in justifying a variance from the 1-mg/L effluent phosphorus standard. A community that wishes to obtain a variance should provide the state with a detailed study plan proposal that would be reviewed, modified as needed, and finally approved by the Vermont Department of Water Resources and Environmental Engineering. It should be noted that this department may require additional personnel and a board of advisors who can critically review these proposals to ensure that information therein is adequate to serve as a basis for granting the variance.

Combined Sewer Overflows

Another source of phosphorus that should receive high priority in a control program is combined sewer overflows. Many communities in Vermont, like other older sections of the United States, have combined sewers in which stormwater and domestic wastewater are carried in the same pipe. To protect the pipe and other structures, the combined sewers have diversion works that, when the flow becomes great enough to risk the structures' safety, route the excess stormwater and sewage to a nearby water course. These combined sewer overflows (CSOs) are recognized as important sources of contaminants that can pollute local waters. Several years ago, the USEPA implemented a program for eliminating all CSOs, which basically meant stormwater and sanitary sewerage systems were separated. It was soon found, however, that the elimination of combined sewers in many communities would be prohibitively expensive. As a result, the USEPA abandoned this approach. However, this change in policy was not based on a finding that CSO's are not significant sources of contaminants which could impair beneficial uses of water. It is now generally recognized that, rather than adopting a blanket policy of CSO elimination as USEPA originally proposed, a site-specific evaluation should be made to assess a particular CSO or group of CSOs' effect on local water quality.

For Vermont's revised phosphorus management strategy, these site-specific evaluations are recommended. Of particular concern for CSO discharges are the nearshore waters close to the point of discharge. Again, the burden of proof (using the methodology outlined in Chapter 3) should lie with the community that does not wish to spend money to separate its sanitary sewers from its stormwater drainage system. If the community can demonstrate that its CSOs are not adversely affecting beneficial uses of the water body receiving them, there should be no need to alter the CSO discharges. Without such proof, the state should require the community to either eliminate CSO discharges or treat them in a way that maintains the receiving waters' quality.

It may be possible to control algal-available phosphorus in CSOs by adding alum (aluminum sulfate) at a rate of several hundred milligrams per liter when discharge occurs. Although this approach would be expensive for some communities, it is far less costly than separating the combined sewers and is highly effective in converting the CSO's available P to unavailable forms. This practice would not have a detrimental impact on fish and wildlife in the receiving water unless the stream has little or no alkalinity or the pH is 5.5 or lower. Most streams should have adequate buffer capacity to prevent the aluminum from going into solution. In Vermont, some streams may fall below this level. For these reasons, if alum addition to CSOs is being considered, site-specific water quality should be studied to ensure that there will not be detrimental impacts or violation of stream standards.

It is important to emphasize that Vermont should not adopt across-the-board requirements for alum treatment of all CSOs. As with the other phosphorus sources discussed, a site-specific evaluation should be conducted with enough technical credibility to prove that phosphorus control is unnecessary for a particular CSO. CSO and domestic wastewater treatment plant bypasses should be treated the same, whether they are located on a lake or are discharged to a tributary. For CSOs entering off-lake surface waters, the amount of available phosphorus ultimately reaching the lake should be assessed.

Urban Stormwater Drainage

The runoff from community streets presents a potential source of phosphorus that must be considered in any eutrophication management program. Large amounts of the phosphorus in urban stormwater have been found to be particulate forms.[87] Only about 20 percent of these particulates is available to support algal growth.[88] This percentage has been found to be essentially the same for urban stormwater drainage from several communities. Therefore, this number should be reliable enough for Vermont to use in developing phosphorus management strategies.

Should urban stormwater drainage appear to be a major source of available phosphorus, possible control strategies should be evaluated. It is important that the control program focus on available phosphorus, not on total phosphorus. In the past, the USEPA and other agencies often have focused urban stormwater drainage programs on removing relatively large particles that can be readily settled in a short-residence-time detention basin. Removing particulate P this way can be effective for controlling a lake's nearshore siltation due to urban stormwater runoff, but it has little or no impact

[87] G. F. Lee, et al. (1980).
[88] W. F. Cowen and G. F. Lee, "Phosphorus Availability in Particulate Materials Transported by Urban Runoff," *Journal Water Pollut. Control Fed.*, Vol 48 (1976).

on the water body's quality since it is the dissolved, finely divided particulate forms that are of primary concern.

As with CSO treatment, about the only potentially effective method of controlling phosphorus in urban stormwater drainage is by adding alum to drainage waters at the discharge point. This approach increases the suspended solids load of the drainage which, in turn, affects turbidity and sediment accumulation. Rarely, however, would either the turbidity or sediment accumulation adversely affect the receiving waters' quality, since urban stormwater drainage already carries a high suspended particulate load. The alum dosage needed to effectively tie up the available phosphorus in urban stormwater drainage must be evaluated on a site-specific basis, but can be expected to be on the order of several hundred milligrams per liter.

Agricultural Sources

Factors that can affect phosphorus loading from agricultural areas are: soil erodibility, drainage area, physiography, soil clay content, proportion of land in row crops, dairying operations, and fertilizer use. However, only two types of agricultural runoff must be considered in developing a phosphorus management strategy: (1) runoff from crop-producing lands to which phosphorus has been added as inorganic fertilizers, and (2) animal manure disposal. With respect to the latter, Vermont was a pioneer in identifying the problems associated with spreading animal manure on frozen soil. Croft has described measures taken to control nonpoint sources of phosphorus from agricultural wastes in the Lake Parker drainage basin of Orleans County, VT.[89] Federally assisted, long-term contracts were made with farmers to provide manure storage, runoff diversion, grass filters, and other runoff control measures. In the two seasons after controls were implemented, the mean spring chlorophyll-*a* concentrations appeared to be declining.

It is important that manures not be spread on frozen soil. This practice can greatly change the amount of phosphorus exported from these soils. Some years ago, Vermont, Wisconsin, and other states with large dairy industries adopted procedures to help farmers construct manure-holding tanks to store the winter's manure until spring thaw, when it could be spread without greatly increasing the runoff P. This approach should be adopted uniformly at all Vermont watersheds for which the drainage from manured lands enters a lake, either directly or via a river system. As with other sources, the control program's potential effectiveness should be evaluated to ensure that any spending will greatly improve water quality.

Inorganic phosphorus fertilization of lands often is alleged to be an important source of phosphorus that causes eutrophication. Typically, except under unusual circumstances, inorganic fertilizer phosphorus contributes little to eutrophication of natural waters. Most of this phosphorus becomes bound within the soil and is not transported in surface runoff, except during heavy erosion. In general, however, erosion-associated phosphorus is largely unavailable to support algal growth. Site-specific evaluations must be made to determine if runoff from an agricultural area represents a potentially important source of available phosphorus for a water body.

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[89]R. J. Croft "A Project to Manage Agriculture Wastes Has Improved the Quality of Vermont's Lake Parker," *Perspectives on Nonpoint Source Pollution* (Kansas City, MO, May 1985).

Sediment-borne and water-soluble phosphorus from agricultural lands in northeastern Nebraska have been studied before and after installing erosion control structures.[90] In both cases, most nutrient losses were associated with eroding sediment, not with fertilizer use. Soluble, inorganic phosphorus concentrations were lower with controls but was proportional to eroded sediment concentration. In another study, 60 percent of the total phosphorus in Ontario agricultural watersheds was from eroded sediment material.[91]

Because phosphorus runoff from lands is difficult to measure, it is suggested that a mass balance approach be used in which the amount of phosphorus transported past a particular location on a river is determined through stream gauging and periodic sampling. (The procedures described in Chapter 3 should be used.) The amounts of phosphorus derived from point sources in the watershed upstream from the gauging-sampling station should be measured or estimated. The difference between this amount and the total load measured at the gauging station can be used to develop a phosphorus export coefficient for the watershed. The amount of agricultural and forest land within the watershed should then be estimated and forested land should be assigned export coefficients of approximately 0.005 g P/m^2/year. From the forested area and this export coefficient, it will be possible to estimate the total amount of phosphorus from this area that would contribute to the river at the gauging station. This amount, coupled with the point source estimates and measured loads, will make it possible to develop an export coefficient for the agricultural lands. If this coefficient is unusually high (considerably above 0.05 g P/m^2/year), then site-specific studies should be done to learn why these agricultural lands are contributing such large amounts of phosphorus. These studies clarify what the sources are and held in determining what can be done to minimize phosphorus runoff from these lands.

Optimizing Control Measure Combinations

Some models only predict water quality changes given a change in phosphorous loading, whereas others can be coupled with cost-benefit studies to determine the most cost-effective combination of eutrophication control measures. The HSPF model used for the Occoquan River basin (discussed in Chapter 4) to evaluate the trade-offs between point and nonpoint source controls is an example of the first type. A model that included a cost-benefit analysis was used to determine how to control nearshore nuisance growth of attached algae (*Cladophora*) at Harbor Beach, MI (Lake Huron).[92]

In the Lake Huron study, the options were: (1) remove phosphorus from source sewage treatment plant discharges, (2) construct off-shore point source discharges, and (3) a combination of the two methods. The cost-benefit analysis was coupled with a dynamic model that was calibrated and verified specifically for the Harbor Beach site. The model contained a transport submodel for soluble reactive phosphorus transported into and exchanged between shoreline model cells and in-lake water and a kinetic submodel that dealt with the biochemical processes associated with *Cladophora* growth.

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[90] J. S. Schepers, "Water Quality from Erosion Control Structures in Nebraska," *Journal of Environmental Quality*, Vol 114, No. 2 (April-June, 1985).

[91] N. A. Berg, "Control of Phosphorus from Agricultural Land in the Great Lakes Basin," *Phosphorus Management Strategies for Lakes*, (Ann Arbor Science, 1980).

[92] R. P. Canale, et al., "Optimal Cost Control Strategies for Attached Algae," *Journal of Environmental Engineering, ASCE*, Vol 106, No. 9 (December 1983).

Results showed that reducing sewage treatment plant effluent to 1 mg P/L combined with offshore discharge (dilution) represented the most cost-effective management strategy.

A linear programming approach has been used to determine how cost-effective various combinations of phosphorus control measures in three stages would be for point, rural nonpoint, and urban nonpoint sources for 11 major basins of the Great Lakes.[93] The program selected the most cost-effective combination of control measures to achieve the target concentration for each lake. It was found that 83 percent of the objectives could be achieved by spending 23 percent of the originally stated cost. The program contained two constraints: (1) maximum phosphorus loading reductions achievable in a given basin and (2) the reduction in loading needed to achieve the target P concentration in each lake. The controls' cost-effectiveness was measured by weighing improvements by a factor reflecting basin population and miles of shoreline.

Another study that evaluated and compared phosphorus control measures for the Great Lakes has concluded that removing phosphorus from municipal wastewater should receive highest priority.[94] This approach would be cost-effective and would substantially reduce nutrient load to the Great Lakes. Controlling urban runoff was not shown to be cost-effective. Except for Lake Erie, phosphorus control from rural lands was shown to be cost-effective, but would not greatly reduce phosphorus load.

In other work, the USEPA Algal Bloom Expert Panel has recommended eutrophication control strategies for the Potomac estuary.[95] The Panel conducted a cooperative investigation and modeling study to determine the cause of widespread, severe algal bloom that occurred in the Potomac estuary from August to September 1983. The recommended control strategies, based on a determination of suspected causes, included reduction of point-source nitrogen coupled with control of nonpoint phosphorus sources. (The point-source reduction of phosphorus was already being achieved to the extent possible by removal at wastewater treatment plants.) Also recommended was control of internal phosphorus loading by increasing the alkalinity of sewage treatment plant discharges, since it was hypothesized that a high pH and aerobic conditions reduce the phosphorus sorption capability of sediment hydroxides.

Although the studies just described always state their conclusions with regard to the "best" management options for attaining water quality goals, it is often unclear if or how the results were applied to policy formation. Given these examples, an optimum combination of eutrophication control measures could be determined for the State of Vermont with regard to Lake Champlain. As Henson [96] noted, in some basins, further loading reductions are needed beyond what could be attained by the P detergent ban and the 1-mg/L effluent level at sewage treatment plants. Other measures that could be considered are: requiring selected wastewater treatment plants to meet discharge limits of lower than 1 mg P/ L, control of CSOs, seasonal effluent limits at wastewater treatment plants, low flow augmentation to prevent conditions favoring algal growth,[97]

[93]S. C. Chapra, H. D. Wicke, T. M. Heidtke, "Effectiveness of Treatment to Meet Phosphorus Objectives in the Great Lakes," *J. Water Pollution Control Federation*, Vol 55 (January 1983).
[94]T. M. Heidtke, et al.
[95]R. V. Thomann, et al., *The 1983 Algal Bloom in the Potomac Estuary*, (Potomac Strategy State/USEPA Management Committee, March 1985).
[96]E. B. Henson, et al.
[97]C .J. Velz, *Applied Stream Sanitation* (John Wiley and Sons, Inc., 1984).

and improved manure management. The optimal combination would be determined separately for each subbasin, taking into consideration target water quality conditions in the bays, control effectiveness, predicted impact of the controls, and cost.

Summary of Policies in Other Basins

Results of a literature search and telephone calls to regulatory and water planning authorities showed that for only one basin was the distance upstream of a point source take into consideration. This was a specific incident rather than a state policy, however. The city of Spokane, WA, was required to remove P at its wastewater treatment plant. An environmental impact study was done which measured the attentuation losses of P discharged from sources 50 miles upstream. It was determined that the contribution of these upstream sources was not significant enough to warrant requiring them to remove P.

The literature search identified studies pertaining to modeling of receiving waters, determining the amount of available P in agricultural runoff, and removal and control techniques for point and nonpoint sources, as discussed elsewhere in this report. The telephone calls were aimed at organizations which are familiar with the most recent studies and aware of activities in several drainage basins. Contacts included: the EPA Great Lakes Program Office, the EPA Chesapeake Bay Program Office, the Potomac Strategy State/EPA Management Committee, the Army Corps of Engineers Buffalo District, the EPA laboratories in Athens, GA, and in Corvallis, OR, the Pennsylvania Academy of Science, the Army Corps of Engineers Waterways Experiment Station, the Northern Virginia Planning District, the New England Water River Basin Commission, the States of Maryland, Virginia, and Pennsylvania, and EPA Regions 6, 7, 9 and 10.

In the Great Lakes region, P removal requirements at sewage treatment plants are basin-wide. In other areas, water quality in the immediate receiving water body determines whether P removal is indicated. In many cases, effluent limits are stricter due to site conditions. It has been recognized and discussed that (1) some P may be lost in transport from an upstream source to the bay and (2) not all the P discharged from a point source may be available for algal growth, but state policies have not been affected. Based on the findings of this investigation, it may be said that the State of Vermont's giving credit to off-lake sources is unique.

6 DETERGENT PHOSPHATE

The April 1977 amendments to Vermont's Water Pollution Control Legislation include a prohibition of the sale of household laundry detergents that contain phosphorus in amounts greater than trace levels--the so-called "phosphorus detergent ban." According to a March 1981 report to Vermont, the detergent phosphate ban resulted in a 40 percent reduction in effluent phosphorus concentration from selected municipal wastewater treatment facilities.[98] This value was derived by comparing wastewater treatment plant operations data for the 1977 preban year with those from the 1978-1979 postban period. Some variability was found among treatment plants in pre- and postban phosphorus concentrations, with 1979 data generally showing higher P concentrations than those of 1978 (i.e., the first year of the ban). Detailed data were not presented in that report, but it is suspected that the situation in Vermont was similar to that in Michigan and Wisconsin.

The 1978 detergent phosphate ban in Michigan resulted in about a 23 percent decrease in phosphorus load to the state's wastewater treatment plants.[99] Similar trends have been seen in Wisconsin. The reason Vermont shows a much higher percentage phosphorus reduction with enactment of the detergent P ban than has been found in other states is unclear. The study methods in the two areas were not the same, so the results are not directly comparable, although it would be expected that the preban detergents sold in all states would be similar.

Another study has presented a detailed analysis of data obtained on phosphorus load changes associated with Wisconsin and Michigan detergent P bans.[100] These data showed great variability in pre- and postban P loads to wastewater treatment plants. Some treatment plants even showed an apparent increase in concentration after the ban compared to preban levels; others showed little or no change, with others showing some decrease. Clearly, seasonal and year-to-year variations in P loads make it difficult to determine changes at wastewater treatment plants that result from a detergent phosphate ban. A detailed evaluation of Vermont's data probably would result in the same conclusion.

For water bodies receiving domestic wastewater treatment plant effluents that are not undergoing P removal, site-specific evaluations should be made to determine what impact, if any, the detergent P ban had on phosphorus loads to the state's water bodies. These effects could be assessed using the Vollenweider-OECD eutrophication modeling approach. Several years of pre- and postban data would be needed to document the impact properly. It is likely that, as a result of the P reductions in U.S. detergent formulations since the mid-1970s, the detergent phosphate ban in Vermont today is

[98] State of Vermont, Department of Water Resources and Environmental Engineering, Water Quality Division, *Special Report to the Vermont General Assembly on the Effectiveness of the Phosphorus Detergent Prohibition in Household Cleansing Products and Compliance with a 1.0 Milligram Per Liter Discharge Limitation* (March 1981).

[99] J. H. Hartig and F. J. Howath, "A Preliminary Assessment of Michigan's Phosphorus [Detergent Ban," *J. Water Pollut. Control Fed.*, Vol 54 (1982).

[100] P. M. Berthouex, K. Booman, and L. Pallesen, "Estimating Influent Phosphorus Loading Shifts Caused by Bans on Phosphate Detergents," *Proc. of the International Conference on Management Strategies for Phosphorus in the Environment* (Selper Ltd., London, 1985).

reducing phosphorus loads to the state's waters by about 20 to 25 percent when domestic wastewaters have no P removal. Smeltzer [101] took the detergent P ban into account in his loading calculations. Any benefits of maintaining a detergent P ban with the recommended across-the-state 1 mg/L P limitation in domestic wastewater treatment plant effluent should be examined for cost-effectiveness. Henson [102] also looked at the effectiveness of both a detergent P ban and P removal at treatment plants.

Chapter 4 discussed the relationships between changes in phosphorus loads to a lake or reservoir and the associated changes in eutrophication-related water quality. Recall that at least a 20 percent change in P load is necessary before most of the public would see a change in water quality, and that more generally, a 30 percent decrease would be needed. Figure 11 shows the detectable water quality changes in a water body based on a relationship between total annual phosphorus load added from domestic wastewater sources and the amount of this phosphorus that is derived from detergent formulations. This relationship shows that, if 40 percent of the P in Vermont's domestic wastewaters is due to detergents, a detergent P ban will visibly improve eutrophication-related water quality only when 50 to 60 percent of the total P load to the water body is from domestic wastewaters that have no P removal. In reality, few water bodies receive this high a domestic wastewater phosphorus input. It is also clear from Figure 11 that with the probable situation in Vermont, where the detergent P ban is reducing domestic wastewater phosphorus by 20 to 25 percent, such a ban is having no discernible impact on the quality of may water bodies. This result has been concluded because a greater P load change than is possible through detergent P bans is required to improve eutrophication-related water quality.

Another factor to consider in evaluating the potential impact of detergent P bands is that the condensed phosphates used in detergent formulations can undergo enzymatic hydrolysis and, under certain conditions, form into unavailable forms of phosphorus. If approximately half the total domestic wastewater P derived from detergent formulations is converted by hydrolysis into permanently unavailable forms, it is likely that no waters in the state will have quality improved due to a detergent phosphate ban.

It is important to emphasize again that, in analyzing the impact of a phosphorus load reduction from any source, critical areas of the water body must be considered (e.g., nearshore regions, beaches, bays). Each area must be analyzed separately to ensure that a particular load change which may have no impact on overall water quality at the lake or reservoir, cannot have localized impact on the water quality in a critical area. This situation could occur because of limited mixing between the particular part of the lake and the main body of the lake. Again, in assessing the impact of altering the P load a certain amount though a certain management practice, site-specific evaluations should be made.

It may be appropriate for Vermont to reexamine the detergent phosphate ban as part of revising its phosphorus management strategy. This information will ensure that the ban is still producing the net benefits to the public as originally planned.

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[101] E. Smeltzer, "Phosphorus Removal Policy" Memorandum to R. Czaplinski, State of Vermont, Agency of Environmental Conservation, Department of Water Resources and Environmental Engineering, Montpelier, VT (21 Sept 83).
[102] E. B. Henson, et al.

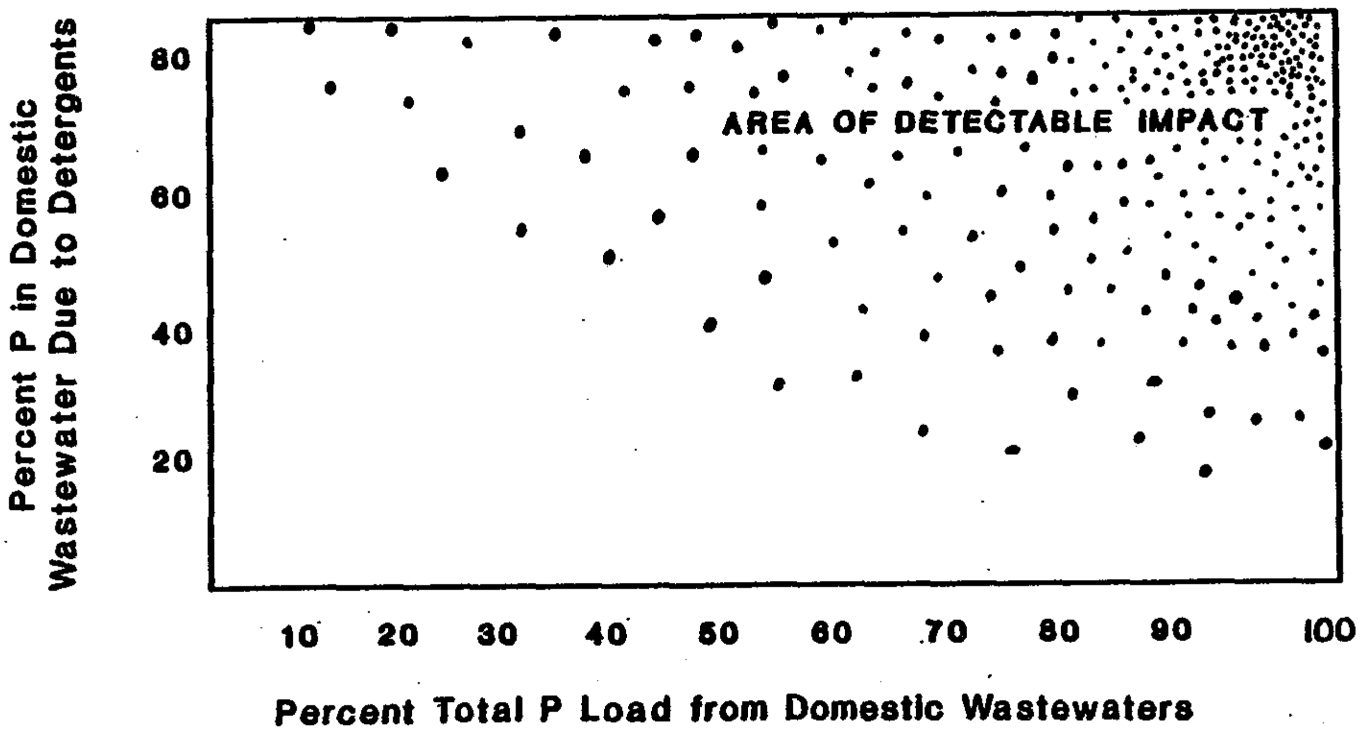

Figure 11. Impact of detergent P on detectable changes in eutrophication-related water quality. A 20 percent change in P load to a lake is necessary to produce a change in eutrophication-related water quality that is just perceptible. The density of dots reflects the degree of public perceptibility of change in water quality resulting from a detergent P ban. The figure is read as follows: if 40 percent of the P in domestic wastewaters were due to detergents, and if the domestic wastewater discharge represented 30 percent of the total P load to the lake, the removal of detergent P would result in no detectable change in eutrophication-related water quality of the water body. If the discharge represented 60 percent of the total P load to the lake, removal of detergent P would result in an improvement in water quality that some would see. An even greater improvement would result from the removal of detergent P if the domestic wastewaters represented 100 percent of the total P load to the lake. (Source: W. F. Lee and R. A. Jones [1985b]. Used with permission.)

7 AVAILABLE PHOSPHORUS IN DOMESTIC WASTEWATER EFFLUENTS

In the past, the phosphorus present in domestic wastewaters was considered to be 100 percent available to support algal growth, so that P loads from such sources were based on total P measurements. Today, however, with increasing numbers of treatment plants removing phosphorus by adding iron or alum salts, much of the phosphorus in domestic wastewater effluents may be in forms unavailable to support algal growth in the receiving waters. As already discussed, iron and aluminum phosphate species are unavailable forms. As reported recently,[103] this finding has not been true for iron or aluminum phosphate species added to soils; terrestrial plants apparently can extract the phosphorus in these forms, whereas algae cannot. It is unclear whether attached aquatic macrophytes can use these forms. Normally, however, the low density of iron and aluminum phosphates, coupled with their fine particle size, would make them settle in a lake at water depths greater than those at which macrophytes typically grow.

With this knowledge, political bodies considering phosphorus management strategies that require communities to reduce their domestic wastewater treatment plant phosphorus to a concentration below 1 mg/L should clarify legislation to mean algal-available P, rather than total P. Several years ago, U.S. and Canadian water quality managers for the Great Lakes adopted regulations requiring that wastewater treatment plants located in some parts of the Great Lakes basin filter their effluents to achieve the 0.5 mg/L P effluent standard. The appropriateness of this approach was questioned,[104] in that filtering the effluent would primarily remove particulate iron and aluminum phosphate species, which are largely unavailable to support algal growth in the receiving waters. The discharge would then contain a higher percentage of phosphorus in a form readily available for algal growth. Thus, the benefits may not be as great as might be expected. That means large sums of money could be spent to control, without improving, eutrophication-related water quality. (See **Costs and Policy Considerations.**) Other studies have shown that, for several treatment plants in the Great Lakes basin, only about 20 percent of the particulate phosphorus in the domestic wastewater effluent of treatment plants removing phosphorus to 1mg/L was available to support algal growth.[105] This figure supports previous concerns that the phosphorus management strategy adopted for a region focus on algal-available P as determined from algal bioassay techniques rather than on total P concentrations.[106]

Biological Removal

A eutrophication control strategy receiving recent attention is biological phosphorus removal.[107] This technique has been known for many years, but is only now beginning to receive widespread use. Advantages to removing P by this approach include reduced sludge handling problems. However, some important questions have been raised

[103] B. Hansson, "Practical Experiences and Official Tests with Chemically Precipitated Phosphate Sludges in Swedish Agriculture," *Proc. of the International Conference on Management Strategies for Phosphorus in the Environment* (Selper Ltd., London, 1985.)

[104] G. F. Lee, et al. (1980).

[105] T. C. Young and J. V. DePinto (1982a and b).

[106] G. F. Lee, et al. (1980).

[107] J. N. Lester and P. W. Kirk (Eds.), *Management Strategies for Phosphorus in the Environment* (Selper Ltd., London, 1985).

about the algal availability of effluent phosphorus from biological treatment plants compared with that from treatment plants using chemical removal. The particulate phosphorus in biological treatment plants may have much greater algal availability; studies are needed to determine if this is true.

Although it typically may not be justifiable to filter chemically treated effluents, filtering those from biological treatment plants may be warranted. Site-specific evaluations should be made to determine the best approach for limiting phosphorus from biological treatment plants below that attainable using this technology alone. A recent study indicated that biological phosphorus removal has considerable merit; however, it recommended that all biological treatment works include chemical treatment capacity for use when the biological processes are not functioning properly.[108]

The Republic of South Africa and other groups have set limitations for soluble orthophosphate effluent P in both biologically and chemically treated domestic wastewaters. (Some treatment plant operators in South Africa have learned that they can more easily "comply" with "soluble P" effluent concentrations by using smaller pore-size filters for the analysis of soluble P.) Although neither soluble P nor total P-based effluent limitations would be technically valid criteria for defining treatment strategies, for plants using chemical treatment, soluble ortho P is probably a better estimate of available P in the effluent than total P; for biological treatment plants, until demonstrated otherwise, total phosphorus as defined by persulfate digestion is a better estimate of algal available P in the effluent than soluble ortho P.[109]

Some European operators have, through careful control of the phosphorus removal plant, obtained an effluent with 0.05 mg/L P.[110] In situations for which most of the phosphorus in a particular water body comes from domestic wastewaters, water quality can be much improved by treating the effluent to this degree. This same result was seen in a study of an electric-generating station cooling impoundment.[111] This study evaluated what degree of domestic wastewater treatment would be necessary to use the effluent as the sole source of water for this impoundment to allow its use for public recreation. Based on the morphology and addition rates of makeup water needed to keep the reservoir full, it was found that a mesotrophic water body (i.e., planktonic algal chlorophyll concentrations less than about 10 μg/L) could be developed if the makeup water were treated to achieve a P concentration of 0.1 mg/L. The domestic wastewater had an average P concentration of about 7 mg/L. Thus, a phosphorus removal plant was designed and operated to achieve the 0.1 mg/L effluent P level; water quality in the reservoir was quite close to that predicted based on the Vollenweider-OECD model.

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[105]E. F. Barth, "Current Status of Phosphorus Removal and Nitrification in the USA," *Proc. of the International Conference on Management Strategies for Phosphorus in the Environment* (Selper Ltd., London, 1985).

[109]APHA.

[110]J. N. Lester and P. W. Kirk.

[111]R. A. Jones and G. F. Lee, "Development of Water Quality Management Program for the Rawhide Electric Generating Station Cooling Impoundment: A Domestic Wastewater Reuse Project," *Proc. of Water Reuse in the Future*, Symposium II (American Water Works Association, 1982b).

Costs and Policy Considerations

The tendency in some situations is to adopt set limits for effluent phosphorus to well below 1 mg/L. There are two factors to consider. One is that in attaining a level below 1 mg/L, filtration is often necessary, leaving a higher percentage of algal available P in the discharge, as discussed earlier. The other factor is cost effectiveness of achieving this low level. In general, phosphorus removal to 0.2 mg/L is about four times as expensive as the 1-mg/L level. For large treatment plants, however, this cost may be relatively minor when computed on a per capita per day basis. It may represent only a few cents per person per day for combined capital and operating costs. P effluent limits below 1 mg/L may be appropriate for large treatment plants discharging directly to the lake or a bay such that the phosphorus would be in a readily available form for algal growth (e.g., St. Albans).

In assessing the need for phosphorus removal at a domestic wastewater treatment plant, it is important to determine if seasonal phosphorus control programs can be implemented. This option is being used in the Chesapeake Bay drainage basin and has been suggested for the Lake Erie basin. For water bodies that have hydraulic residence times on the order of a year or so and are reasonably well mixed, year-round P control programs are necessary. However, for water bodies that have short residence times (on the order of 2 months or less) and that tend to be well flushed during the heavy spring flows, seasonal P removal programs may work. To develop the most technically valid, cost-effective approach for managing excessive fertilization, site-specific analysis should be done to determine if a particular domestic wastewater P discharge impairs a water's beneficial uses. Again, the available phosphorus load change is the important variable to study.

8 PHOSPHORUS MANAGEMENT RECOMMENDATIONS FOR VERMONT

Vermont's surface waters are being excessively fertilized by phosphorus derived from residents' activities. The primary source is domestic wastewaters, but major contributions are from urban and agricultural lands. From this review, it is concluded that Vermont should critically review its phosphorus management strategies to bring them more in line with technical information available on how phosphorus discharges can affect surface water quality.

Although potentially large amounts of phosphorus discharged to a river can be converted to forms unavailable for supporting algal growth, the extent to which this occurs is highly river-specific. Thus, a site-specific evaluation is always needed to ensure that phosphorus discharged to a river from an off-lake community remains in an algal unavailable form. It is recommended that Vermont adopt a phosphorus management strategy that would require all communities with domestic wastewater flows above about 0.2 mgd to use a phosphorus removal treatment that results in a 1-mg/L total P effluent concentration. This approach is consistent with current policies in other drainage basins with regard to the assumption that all phosphorus reaches the receiving water body, as well as with the practice of granting exemptions to small plants due to high per capita removal costs. Small communities should be able to receive a variance from this requirement provided they can prove to the state that an exemption (1) will have no major impact on downstream lake or reservoir water quality and (2) would represent an excessive financial burden to the community.

The state, in cooperation with the communities affected by this new phosphorus management strategy, should conduct field studies to determine how much algal available phosphorus added to local rivers from point and nonpoint sources is converted to permanently unavailable algal forms of phosphorus during transport from the discharge point to the lake. These studies should be funded primarily by the communities wishing to receive a variance from the 1 mg/L domestic wastewater effluent P standard.

The revised phosphorus management strategy also might include controls for algal available phosphorus from agricultural drainage, with particular emphasis on animal manure. It is recommended that the practice of spreading manure on frozen soil be prohibited unless proven to have no adverse effect on downstream lake or reservoir water quality.

Adopting these recommendations will provide Vermont with a more technically valid, cost-effective approach for managing the excessive fertilization of its surface waters than is being practiced today.

9 CONCLUSIONS AND RECOMMENDATIONS

Current information has been reviewed on the availability of phosphorus, discharged from offlake point and nonpoint sources, for supporting phytoplankton growth in receiving water bodies. Eutrophication management strategies have been presented with respect to phosphorus control. Investigation of policies in other watersheds has revealed that no consideration was given to transport and attenuation losses when developing P control strategies. This information has been used to recommend guidance on approaches to developing an updated phosphorus management strategy for the State of Vermont.

It was found that, although major amounts of phosphorus added to a river may be converted from algal-available forms to unavailable forms, the extent to which this occurs is highly river-specific. Thus, across-the-board requirements for domestic wastewater treatment to a specific phosphorus concentration may not be justified in all cases—especially for some smaller communities' discharges from which phosphorus reaching a lake or reservoir in the available form is too little to be discernible and for which P removal costs would be unacceptably high. In contrast, too few controls may govern other potential phosphorus sources, such as those related to some agricultural practices.

Site-specific field studies should always be conducted to determine how much algal-available phosphorus entering a river from point and nonpoint sources eventually reaches a main water body in an available form. It is also essential to measure the impact on important uses of the lake or reservoir and its nearshore waters (e.g., swimming, fishing). Eutrophication management strategies for Vermont should include these activities:

1. Limit phosphorus in effluents to a 1-mg/L total concentration for all domestic wastewater treatment plants greater than 0.2 mgd.

2. Allow exemptions to this rule if a community proves the restriction will not visibly improve water quality and will impose an excessive financial burden.

3. Establish controls on the storage and handling of animal manure—in particular, prohibit farmers from spreading it on frozen ground.

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